Polychlorinated Biphenyls (PCBs) - Fate and Effects in the Canadian Environment

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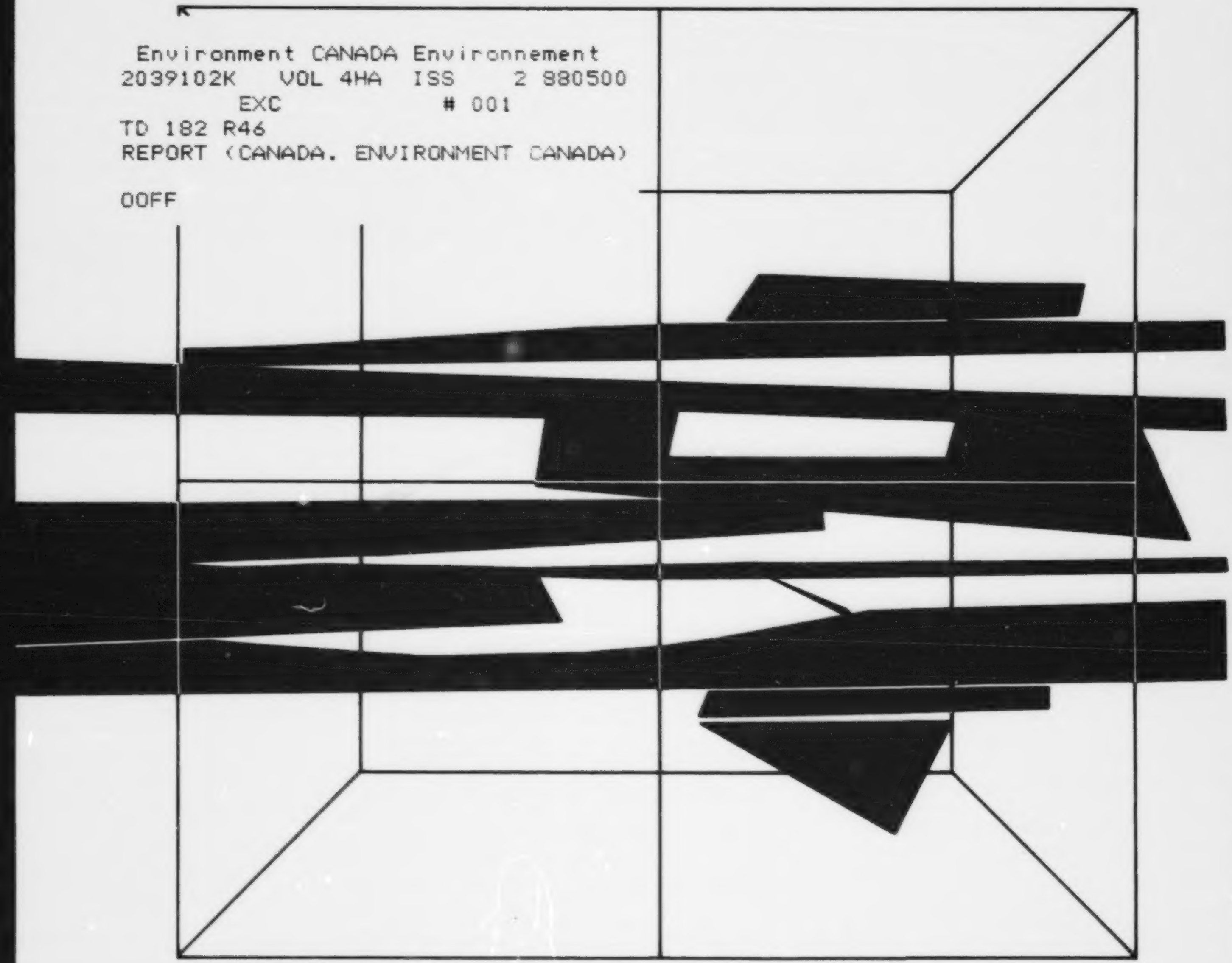

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POLYCHLORINATED BIPHENYLS (PCBs) - FATE AND EFFECTS IN THE CANADIAN ENVIRONMENT

prepared for

Toxic Chemicals Steering Committee
Canadian Council of the Resource and Environment Ministers

by

W.M.J. Strachan
Conservation and Protection
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May 1988

PREFACE

This document was commissioned by the Toxic Chemicals Steering Committee (TCSC) of the Canadian Council of Resource and Environment Ministers (CCREM) following the spill of polychlorinated biphenyls (PCBs) on the Trans-Canada Highway near Kenora, Ontario in April, 1985. This assessment of fate and effects of PCBs in Canada formed one part of the PCB Action Plan implemented by CCREM.

This report was prepared by a single author with input from a large number of federal, provincial and other scientists and personnel. Although the data and information are technically correct, the conclusions, recommendations and opinions expressed in the report reflect those of the author and not necessarily those of Environment Canada or the federal government.

The report has undergone some limited editing as well as translation for publication as an Environment Canada Report. However, the data and information have not been updated and readers should keep in mind the fact that more scientific and technical information on PCBs has been published since this report was completed.

READERS' COMMENTS

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ABSTRACT

This report gives an environmental assessment of the hazards posed by polychlorinated biphenyls (PCBs). It is an interpretive presentation of data on exposure and effects which were available after publication of the original "Report of the Task Force on PCBs" (Task Force, 1976) prepared jointly by Environment Canada and Health and Welfare Canada. A comparison of expected exposure levels and their effects is presented, as well as recommendations for control measures.

RÉSUMÉ

Ce rapport évalue les dangers que posent les BPC (biphényles polychlorés) dispersés dans l'environnement canadien. Il s'agit d'une interprétation des données sur l'exposition aux BPC et ses effets, ultérieures à la publication, en 1976, du rapport du Groupe de travail sur les BPC préparé par Environnement Canada et Santé et Bien-être social Canada. Le lecteur y trouvera des comparaisons entre les niveaux d'exposition vraisemblables et les effets probables, ainsi que des recommandations visant à renforcer les mesures de contrôle.

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A large number of individuals from Environment Canada, Fisheries and Oceans Canada, and Health and Welfare Canada provided data and reviews, some of which were and are unpublished. Others facilitated by referring the data request to others and by providing review comments. Information was also obtained from some provincial agencies and the U.S. Environmental Protection Agency. The author is extremely grateful to those who provided data and assistance, their names and affiliations follow:

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SUMMARY AND RECOMMENDATIONS

Summary

Review of the data on exposures to PCBs in the Canadian environment leads to the conclusion that there is extensive contamination in most parts of the country with the Great Lakes being an area of particular concern. In this regard, the situation has not changed since these compounds were first examined under the Environmental Contaminants Act in 1976. There has, however, been a reduction in the PCB levels in the Great Lakes biota so that the original disparities between that area and the rest of the country are not so pronounced. National levels in humans, however, appear to have increased somewhat.

Evidence suggests that the atmosphere is a major medium whereby PCBs are transported through the biosphere. The distribution of these chemicals appears to be continental, if not global. While the quantities in the atmosphere may not be large compared with known world losses to the environment, the fluxes between the atmosphere and the water, and possibly the soil, suggest this may be of importance in the overall cycling.

Data were lacking on the expected exposures for humans through food sources. Residue levels for foods outside of the Ontario Region are particularly required as well as levels in plant-related materials from across the country. These information gaps prevent reliable estimates of human exposure to be made although speculation leads to the conclusion that while foods are the major vector for humans, the levels for Canadians are not such as to presently indicate harm. Data for residues in wildlife and its food sources are, for the most part, unavailable.

Effects data for PCBs are almost always carried out on a single species basis. While this may be desirable for evaluating the hazards that may face that particular species, especially if the species is man, it is not sufficiently revealing about the effects on the species in its environment. Some studies in the aquatic field have shown that community response is comparable in sensitivity to responses of sensitive organisms. This type of information would lend greater confidence in assessment conclusions than can presently be accorded.

Some toxicology data is available which shows there are distinctly different effects to be expected depending on the degree and position of chlorine substitution in the biphenyl nucleus. Almost no comparable data exist for congener-specific environmental residue levels which would facilitate an assessment on this basis and hence a truer picture of the hazards of PCBs. Temporal and spatial trends in PCB data have not generally been possible because of lack of a suitable basis for comparison. As a consequence, it is not possible, even eight years after the first regulation on PCBs was promulgated, to state what the effectiveness of that action has been.

Recommendations

The following recommendations are made for monitoring-related activities for Environment Canada - either to perform itself or to promote others into doing. These actions are intended to provide for a better exposure assessment, including trend analysis:

(1) Evaluate new sampling and analysis technology already developed by the department for the determination of residue levels in water and in the atmosphere. Specifically, the APLE sampler (water) and a continuous flow water sampler, an all-weather wetfall sampler and a dichotomous vapour-particulate sampler currently available as prototypes should be examined. Congener-specific analytical techniques should be used and reported.

(2) Determine the importance of water-air interchange in the overall budgets of persistent toxic substances such as PCBs for different waterbodies.

(3) Undertake monitoring programs for PCBs and other substances which are based upon fewer stations with more frequent sampling and of a sort that permit comparability between different parts of the country. Such monitoring efforts should aim at providing comparability over time and consequently should attempt to make evaluations of methodology available whenever there is a change in techniques. These programs should utilize time-integrating samples and, where biota are considered, should preferably be those with a limited forage area. The monitoring should include determination of PCBs in biota at various trophic levels and, where the necessary mandates are multidepartmental, the department should take the lead and establish effective co-ordination of sample collection and analysis.

(4) To the extent possible, the entire department, and other government and non-government agencies, should be encouraged to submit data to a common data base such as NAQUADAT on a timely basis. This should be made mandatory for all departmentally-funded studies. The data base managers should be required to adapt their storage and retrieval formats so as to provide data on various life stages of species as well as the location from which the sample was collected and to make provision for congener-specific data (for PCBs) and similar collective "compounds".

(5) In the Atlantic Region, additional residue data should be sought for fish and for lower trophic levels. There is also a need for data from the Labrador area.

(6) In the Quebec Region, data from the northern areas of the region should be obtained for all sample types. The existing data for the St. Lawrence should be sorted so as to allow appropriate temporal and spatial comparisons.

(7) In the Ontario Region, sampling and analysis should be undertaken on a more regular basis outside the Great Lakes portion of the region.

(8) In the Western and Northern Region, the dichotomy of the high levels found in fish and the low sediment levels in the North Saskatchewan River should be resolved.

The apparent trend to higher levels found in the livers of polar bears in the arctic portion of the region should also be investigated.

(9) In the Pacific and Yukon Region, monitoring in areas other than the Fraser River and on occasions other than spills should be undertaken, and integrated with monitoring programs as indicated in (3). More emphasis should be placed on obtaining environmental data from the Yukon portion of the region.

(10) A one time survey for the presence of PCBs in a wider range of food stuffs than is presently investigated should be undertaken. A particular purpose should include the resolution of whether or not fruits, cereals and vegetables contribute significantly to the Canadian PCB body burden.

In addition to the preceeding recommendations, the following recommendation is made to promote an understanding of the ecosystem toxicological effects of PCBs.

(11) Researchers should be encouraged to undertake community response studies wherever feasible. To date, this type of study has focussed on the lower trophic levels in the marine environment but opportunity should be given for innovative studies with other systems.

There is no recall for a chemical that has truly escaped to the environment. The PCBs that are cycling in the Canadian system at present will continue to do so until nature "removes" them by sediment burial or by degrading them; there is little effective action that can be taken to speed the process. The prudent manager, however, will do all that is possible to prevent the escape of any additional PCBs. Current regulations limit the use of PCBs to existing equipment, largely electrical, and provide for the safe transport of waste PCBs for the purpose of destruction. To protect against further PCB releases, there are few additional regulatory actions that can be taken.

While recommendations on control actions were not explicitly mandated as a topic for this review, they were implicitly so and the opportunity is taken to urge several actions which it is felt will assist in providing longer term environmental protection. These appear as the next two recommendations (the second is contingent upon the success of the first). The final recommendation acknowledges the global aspect of the problem.

(12) Suitable destruction technology should be put in place capable of handling all of the existing stored PCB wastes within a few years. While acknowledging that the primary responsibility for this under Canada's constitutional arrangements lies with the provinces, the department should vigourously promote this. Cost-effective methods appear to be available.

(13) The existing regulations under the Environmental Contaminants Act should be investigated to determine how they can be used to speed up the normal phase-out of the existing equipment particularly that which may still use PCBs as dielectric fluid.

(14) The department and government should consider using all international forums to urge other nations, especially the industrialized ones, to move in the direction of more stringent controls (preferably a total ban) on the use of PCBs and to make their destruction technology and facilities internationally available.

GLOSSARY

COMMON NAME	LATIN NAME
Birds	
American Kestrel	*Falco sparverius*
Common Tern	*Sterna hirundo*
Double-crested Cormorant	*Phalacrocorax auritus*
Great Blue Heron	*Ardea herodias*
Greater Shearwater	*Puffinus gravis*
Herring Gull	*Larus argentatus*
Mallard	*Anas platyrhynchos*
Merlin	*Falco columbarius*
Ring-necked Pheasant	*Phasianus colchius*
Prairie Falcon	*Falco mexicanus*
Bobwhite Quail	*Colinus virginianus*
Razorbill	*Alca torda*
Ringed Dove	*Streptopelia risoria*
Sooty Shearwater	*Puffinus griseus*
Mammals	
Beluga Whale	*Delphinapterus leucas*
Grey Seal	*Halichoerus grypus*
Harbour Porpoise	*Phocoena phocoena*
Harp Seal	*Pagophilus groenlandicus*
Polar Bear	*Ursus maritimus*
Ringed Seal	*Phoca hispida*
Fish	
Arctic Char	*Salvelinus alpinus*
Atlantic Herring	*Clupea harengus*
Brook Trout	*Salvelinus fontinalis*
Brown Bullhead	*Ictalurus nebulosus*
Brown Trout	*Salmo trutta*
Burbot	*Lota lota*
Carp	*Cyprinus carpio*

COMMON NAME	LATIN NAME
Fish (cont'd)	
Channel Catfish	*Ictalurus punctatus*
Chinook Salmon	*Onchorhynchus tshawytscha*
Coho Salmon	*Onchorhynchus kisutch*
Fathead Minnow	*Pimephales promelas*
Gizzard Shad	*Dorosoma cepedianum*
Lake trout	*Salvelinus namaycush*
Largemouth Bass	*Micropterus salmoides*
Largescale Sucker	*Catostomus macrocheilus*
Mosquitofish	*Gambusia affinis*
Northern Squawfish	*Ptychocheilus oregonensis*
Pike	*Esox lucius*
Pumpkinseed	*Lepomis gibbosus*
Rainbow Smelt	*Osmerus mordax*
Rainbow Trout	*Salmo gairdneri*
Sheepshead Minnow	*Cyprinodon variegatus*
Splake (Brook trout/Lake trout hybrid)	
Spottail Shiner	*Notropis hudsonius*
Walleye	*Stizostedion vitreum*
Whitefish	*Coregonus* sp.
White Sucker	*Catostomus commersoni*
Yellow Perch	*Perca flavescens*
Other	
Water Flea	*Daphnia magna*
Fruit Fly	*Drosophila melanogaster*

1 INTRODUCTION

1.1 Background

Polychlorinated biphenyls (PCBs) have been an environmental problem in Canada for a number of years. In 1976, under the impetus of Canada's new Environmental Contaminants Act, this group of chemicals was investigated extensively (Task Force, 1976) and subsequently regulated in 1977. PCBs continue to be observed by departmental and other researchers and, periodically, reports of human exposure either through spills or ingestion of PCB-contaminated fish appear in the press. These occurrences arouse great concern, not only in the public sector as a whole but also in the scientific community. The concern is whether or not sufficiently effective measures have been taken or if further controls on PCBs should be put in place. There is also the fear that little is known about the effects of long-term exposures at environmental concentrations which are generally much lower than those studied in the laboratory.

In May, 1985, this review was requested by the Chairman of the Toxic Chemicals Steering Committee of Environment Canada, Dr. R.W. Slater, Assistant Deputy Minister for the Environmental Protection Service at that time. This request was supported by the then Assistant Deputy Minister for the Environmental Conservation Service, Dr. W.B. Mountain, and the present document is the outcome of their directives.

The terms of reference for the report, as received, were:

1. It should be a critical review of PCBs in the environment and their relevance to environmental and human health; and
2. It should be based upon assessments from the literature and on the author's interpretation of available information.

1.2 PCB Reviews

There have been a number of reviews on the exposures and effects of polychlorinated biphenyls (PCBs). In Canada, a Task Force on PCBs was established under the Environmental Contaminants Act in 1975 and this group reported on the situation in 1976 focussing largely upon exposures that existed at that time (Task Force, 1976). Part of their mandate included evaluating the Environmental Contaminants Act as a means of carrying out a decision of the Organization for Economic Co-operation and Development (OECD) and its recommendations on PCBs (OECD, 1973).

Based upon the Task Force's deliberations and report, the Environment Canada/Health and Welfare Canada Committee of the Environmental Contaminants Act proposed a regulation banning all importation, manufacture and use of those PCBs containing three or more chlorine atoms except for use as a dielectric fluid in transformers and power capacitors (new and used), and except for use in heat transfer and hydraulic equipment (used only). The government passed this regulation in 1977. A subsequent revision of this regulation in 1980 prohibited the use of PCBs in all new equipment as well as use for servicing existing electrical equipment. Additional regulations on release of PCB-contaminated material came into force in August, 1985. The only legal use for PCBs at present is use in existing equipment (mainly in transformers and capacitors as a dielectric fluid). Such equipment has a finite lifetime although some estimates that they can last longer than 35 years have been made.

The purpose of this review was to: update the original Task Force description of the PCB situation; to consider whether or not there has been a decrease in exposures to the Canadian environment; and to suggest additional options that the Government of Canada and Environment Canada, in particular, might take. The data review has focussed upon the post-1976 period although a brief synopsis of the condition at that time is also described. The review also adds material on the effects of PCBs, material which was not included in the original report. There are also comments and recommendations on control measures which are not strictly part of the assigned terms of reference.

1.2.1 The Organization for Economic Co-operation and Development (OECD) and PCBs. One of the earliest motivations for Canadian governmental action on PCBs was the OECD decision of February 13, 1973 (OECD, 1973). Previous to this, the Canadian situation had been summarized in a report by Buxton et al. (1972). The OECD decision recommended that member nations restrict the use of PCBs to dielectric fluids in transformers and capacitors; heat transfer equipment (other than that used for food processing); hydraulic fluids in mining operations; and small capacitors. They further urged nations to: provide a safe means for the disposal of surplus PCBs and PCB wastes; to establish a common labelling system; to develop safe means of transporting such materials; and to limit their exports.

In 1982, the OECD reviewed the PCB situation in member countries from the time controls were placed on PCBs (OECD, 1982). According to the OECD, most member nations had implemented limitations, there was little progress on export controls, and the transport and labelling recommendations had met with uncertain response. They also added that, in their opinion, although the 1980 OECD production of 16 600 tonnes was down 62% from pre-decision production there had been little change in PCB environmental levels. The OECD also noted in particular that only in the Great Lakes had any decline been observed (gull eggs and some fish), all other locations showed either a steady state situation or increasing levels in environmental samples. The OECD stated that the major remaining problem with PCBs was their disposal, the most effective means for liquid PCBs being incineration at temperatures over 1 000 °C with residence times of at least two to five seconds. It was further noted that at the time of the report, Canada had recognized the destruction problem but had not determined their own particular solution. The United States has approved two incineration plants for PCB destruction (Texas and Arkansas) as well as a mobile unit. These incinerators, however, are not available for the destruction of Canadian wastes, and so wastes are being stockpiled pending a "Canadian solution".

Currently, there are recommendations being considered by the OECD, that would phase out all uses of PCBs and only permit their transport for destruction.

1.2.2 **Report of the Task Force on PCBs (1976).** The Task Force responded to the OECD recommendations by noting that Canada was proposing to be more restrictive than the Council decision called for in that only the use of PCBs in electrical equipment was to be permitted. Generally, concerns about labelling, transporting, and disposal have not been so successfully accommodated (OECD, 1982) although recently, considerable efforts have been made on the part of the federal and some provincial authorities in this direction.

The Task Force report described the PCB situation in Canada up to 1975. Total imports of PCBs up until 1974 (none were manufactured in Canada) were listed in this report at 15 775 tonnes, 83% of which had been used for the manufacture of transformers and capacitors -- nominally closed uses as far as environmental release is concerned. A more recent survey reported imports totalling 40 000 tonnes up until 1977 (Env. Can., 1985). The Task Force was unable to indicate with any precision what fate had befallen this material, but perusal of the report leaves little doubt that PCBs which could not be accounted for in current equipment had probably been released to the environment. The only question seemed to be whether or not the PCBs were in a highly mobile phase, as in the atmosphere, or in a relatively immobile one as in a landfill site with a low water table.

The Task Force went to some length to describe the PCB residues in various media in Canada. Herring gull eggs were one such medium and collections had been done which allowed comparisons to be made among the Great Lakes, the St. Lawrence River, the Gulf of St. Lawrence and some Atlantic coast locations. The PCB residues in gull eggs from Lake Ontario exceeded those from the other Great Lakes by three to four times; those from the St. Lawrence River by five to six times; and those from the Gulf of St. Lawrence and Atlantic coast by twenty to thirty times. In the Great Lakes, there were also many situations in which mean levels of PCB residues in certain fish species exceeded food consumption guidelines. Also reported for this part of Canada were instances of sediment contamination.

In the prairie provinces, and even more so for the Pacific coast, there was a paucity of data on PCB residues. Data available, however, indicated that the prairies were receiving these contaminants. The eggs of the Prairie Falcon and the Merlin both indicated PCB contamination although not at the higher levels noted for other species from the eastern parts of the country. This was probably a consequence of the prairie birds' terrestrial food web rather than the gulls' aquatic one. Comparability among species from different regions was not possible because of different migratory patterns and feeding habits as well as physiology and biochemistry.

Considerable data on PCBs in Arctic mammals (seals and polar bears) were available. This information was usually obtained from tissue samples. PCBs were found at lower levels (seals) in the Arctic than concentrations found in similar species from the Atlantic coast. Seals from Hudson Bay/James Bay region, however, appeared to have levels comparable to those from the Atlantic coast. Arctic fish also showed identifiable contamination but at much lower levels than other parts of the country. Again, it was noted that fish (char) in the Hudson Bay area had elevated levels of PCBs compared to the rest of Northern Canada.

Residue levels in the Canadian population and dairy foods (milk) were also presented in the Task Force report. The national average concentration of PCBs in adipose tissue was 0.91 μg/g, with women being less than men by 33% and Ontario being greater than the average by 18%; Manitoba and Saskatchewan were lower than the average by 55%. A summary of these levels as reported by the Task Force is presented in Table 1.

TABLE 1 SUMMARY OF PCB RESIDUE DATA (Task Force, 1976)

	Lake Ontario	Other G. Lakes	St. Lawr. River	Atlantic Prov.	Prairies and B.C.	Arctic Region
			µg/g			
Herring Gull Eggs	108.	48.5	20.7	4.3		
Yellow Pickerel	2.0	0.89	2.0		0.093	
Sucker	1.2	1.7	1.4		0.07	
Halibut				0.36	0.060	
Cod				0.086	0.010	
Flounder				0.12	0.018	
Clam				0.037	0.018	
Char						0.015
Seal Blubber-Harp			9.2	6.2		
-Ringed				35.6(1)		1.2(2)
Polar Bear -Fat						5.5
-Liver						2.7
Prairie Falcon Eggs					0.89	
Merlin Eggs					1.40	
Dairy Milk Fat	0.10					
Human Adipose Tissue	1.1 (4)		0.97(3)	0.73	0.72	

Notes:
(1) excluding a single sample of 3.0 µg/g
(2) mid-range point used for some locations
(3) Quebec
(4) Ontario

2 GENERAL ASPECTS OF PCB EXPOSURE

Polychlorinated biphenyls (PCBs) are in a class of organic chemicals which have a biphenyl nucleus with chlorine substituents. There are varying degrees of chlorination possible for the biphenyl structure but those generally found in samples taken in the environment and commonly used in earlier industrial applications were 40% to 60% or a total of three to seven chlorine atoms per biphenyl structure.

There are many similarities among the properties of the congeners, including those with the degree of chlorination corresponding to the environmental samples. It should always be borne in mind, however, that commercial "PCBs" are mixtures of a number of compounds. The Monsanto Co., which produced more than half the world's production of PCBs and was the sole producer in North America, designated its PCB mixtures as "Aroclors" and added a number, the last two digits of which were the percent chlorine content. Mixtures produced elsewhere were called Kanechlor, Clophen, Fenechlor, and Phenochlor to name the more common trade names used in OECD countries. PCBs which occur most frequently in environmental samples resemble the Aroclor 1254 or 1260 mixtures which contain 54% and 60% chlorine, respectively (four to six and five to seven chlorine atoms per biphenyl nucleus, respectively). More than 100 PCB congeners have been identified in environmental samples; 209 are potentially possible.

PCBs are very stable under most conditions -- both those in which they have been used and in the environment where they are presently found. They were originally used in a wide variety of ways, many of which depended upon their stability, either thermal, electrical or environmental. PCBs were used as dielectric fluids in power transformers and capacitors; as hydraulic fluids; as various specialized industrial oils, plasticizers, and wood sealants; as volatilization suppressants in pesticide formulations, as components in printing ink and carbonless copy-paper; as flame retardants, and probably others. Extensive use of PCBs began in the 1930's and continued in North America until 1971 when the sole manufacturer in the U.S. (Monsanto Chemical) restricted their use to closed applications -- effectively limiting use to electric transformers and capacitors. An estimated 5.7×10^8 kg were used in the United States up until 1970, 3.5×10^8 kg (61%) of which was still in use in 1975. Only 0.25×10^8 kg (4%) of this amount had been destroyed by incineration or environmental degradation, another 0.68×10^8 kg (12%) was estimated to be "free" in the environment, and 1.31×10^8 kg (23%) was located in landfill sites (EPA, 1977). Although there was extensive use of PCBs, they were never manufactured in Canada. The Task Force report on PCBs (1976) describes those uses in some detail. Since 1971, PCBs have been further restricted by legislative or regulatory action in many of the OECD countries including Canada.

Most of the PCBs in North America were used in the electrical power transmission industry and old equipment, complete with remaining dielectric fluids, often ended up as buried material (Nisbet and Sarofim, 1972). Some of the fluids were released to the environment through processing in the paper recycling industry or were applied directly as adventitious components in road oils. Other PCBs, as constituents of various products, ended up incinerated either accidentally at landfill sites or intentionally in solid waste incinerators. Whichever the route, they became an addition to the environmental pool of PCBs. Approximately 16 million kilograms of PCBs are presently unaccounted for in Canada.

2.1 Analytical Aspects of PCBs

Analyses are performed by partially separating the individual PCB-congeners on a gas chromatograph (GC) which is usually equipped with an electron capture detector, a device particularly responsive to high chlorine content compounds. Quantification is done by comparing 20 to 30 of the separated peak heights or areas (detector reponses) to those from a standard mixture with a similar pattern of peaks. Identification from among the various other compounds occurring in the sample is done by removing as many of these interfering substances as possible (the cleanup step) followed by the GC analysis and comparison of retention times on the GC column with times for the standards. Often, a second column with different properties and retention times is used to confirm the identification. Occasionally, confirmation of the identification is done using a mass spectrometer as the detector for the gas chromatograph, but this is usually not possible with most ambient environmental samples because higher quantities of PCBs than those found in most samples are required. At present, realistic levels for reliably quantitating PCBs on a routine basis are approximately 0.01 μg/L for water, and 0.1 μg/g for both sediment and biological tissue; levels down to about 10 to 20% of these are usually reported only as "trace" amounts. Lower levels are possible in research operations but they cannot deal with the throughput of samples necessary for largescale sampling operations.

The fact that PCBs are not a single compound has caused difficulty in analysis and, even today, results in less sensitive analyses for PCBs than for the chemically distinct organochlorine pesticides. The lower chlorinated isomers are more readily degraded; therefore, the pattern most usually observed in the environment resembles the higher chlorinated mixture of congeners and it is usually as these mixtures or as totals that the reports are made. In this report, PCBs refers to PCBs in total and does not distinguish between the congeners unless stated so. At present, there is inadequate information on the environmental exposures of enough individual congeners to warrant undertaking a congener-specific assessment, although effects are often reported for individual congeners. In the future, however, it will be desireable to move in this direction since there is considerable evidence to show that toxicity to many species is dependent upon the PCB structure. Current data are often determined with capillary GC columns and consequently, it is possible to report concentrations for individual congeners rather than the present practice of expressing them collectively as "Aroclor 1254", etc. Mullin et al. (1984) have presented considerable analytical details to characterize the congeners.

2.2 PCB Properties and Environmental Distribution

The structures of the individual PCB components are sufficiently similar that their relative distributions in an ecosystem will be qualitatively similar. They differ somewhat in their susceptibilities to biodegradation and in their solubilities and vapour pressures, all of which will influence the time to achieve an environmental steady-state and cause a preferential removal of the lower chlorinated congeners. In the environment, over time, their properties result in the appearance of a similar composition of congeners as is found in the higher chlorinated original commercial mixtures. For this reason, they are often reported as mixtures; it has little relation to what was originally released.

The low solubilities, especially of higher chlorinated congeners; volatility; and lack of reactivity are three properties of PCBs in particular which are responsible for much of the exposure that environmental organisms experience and the resulting biological effects that are either found or presumed.

2.2.1 Solubility. Solubility is much simpler to define and determine than reactivity or volatility; it is, however, at least as significant in determining the distribution of PCBs. Water solubilities for PCB-congeners have been reported as 0.016 to 640 µg/L for individual congeners with those of higher chlorine content having lower solubilities (Mackay et al., 1983). Commercial Aroclor mixtures with 40 to 60% chlorine content have reported water solubilities of 2.7 to 250 µg/L (EPA, 1980). Potential freshwater levels might approach these values, or even exceed them if complexed with naturally occurring organic matter. Those for marine situations would probably be less. In the real environment, however, saturation concentrations are seldom, if ever, achieved.

Such low solubilities determine the various partitionings that occur in the environment -- to sediments, biota, soils and even the air. There are several empirical equations which relate the equilibrium partition constants (the ratio of the concentrations in the other media to that in water) to water solubility. Means of estimating these and other properties are presented in detail by Lyman et al. (1982). In general, PCBs are strongly bound to organic matter in sediments and soils having partition coefficients for 100% organic matter of 16 000 to 112 000 in these matrices (Karickhoff, 1981). For biota, the partition coefficient is called the bioconcentration factor and is reported as 740 to 4 125 000 (EPA, 1980). It appears to correlate well with the fat or lipid content for fish species (Schnoor, 1982).

An alternative to using the solubility as the prime predictor of partitioning properties, aside from actually determining the property, is the use of the octanol-water partition coefficient. Use of this parameter presumes that octanol is a good surrogate for lipid material. In the case of biota, this is the fat of the organism; for soils and sediments, it is the organic matter. Considerably better predictions of other properties have been obtained using this parameter than with the solubility itself. Predicted values for soil/sediment partition coefficients and for biota are generally in good agreement with experimental values.

2.2.2 **Volatility.** Volatility is much more difficult to estimate than solubility. There is, at present, no accepted method of measuring "volatility". It is generally presumed that it is a property determined not only by the vapour pressure of the PCB mixture (5.5×10^{-6} kPa for 1242 to 16.7×10^{-9} kPa for 1260, Mackay et al., 1983) but also by other factors such as the depth of the water body; the concentration of dissolved PCBs in the bulk water; the air movement (wind); and the turbulence at the air/water interface. Transport of PCBs through the atmosphere via particulate matter, aerosols or vapour phase and its volatilization from soil and plants should also be similarly influenced and equally complex. Qualitatively, the volatility is represented by the vapour pressure of the substance which is used, together with the solubility, to calculate an air/water partition constant (Henry's constant) and hence a rate of loss (Lyman et al., 1982). This rate of loss is also determined by the use of transfer coefficients -- one for the water phase and one for the gas phase. Depending on the value of the Henry's constant, the limiting rate of mass transfer for PCBs occurs in either the gas phase or in the water. There are assumptions that have not been thoroughly tested and to confirm these predictions, accurate measurements of the mass transfer coefficients and the water and air concentrations must be done. There is also the assumption that PCBs can be represented by distinct values for characteristics such as vapour pressure in much the same way as individual congeners. This assumption can only be an approximation and a true understanding of PCB environmental behaviour will require evaluation on a congener-specific basis.

Fluxes of PCBs to the waters of the Great Lakes have been estimated to be the largest input, certainly for the upper lakes (Eisenreich et al., 1981); volatilization has also been estimated (Andren, 1983). It is not known if these waterbodies are currently serving as sinks for or sources of atmospheric PCBs in the region. Wet deposition (rain and snow) has been measured somewhat; dryfall (vapour exchange and particulate) as well as volatilization from water have only been estimated. It must also be assumed that the deposition/volatilization cycle will have a terrestrial counterpart, but this is poorly understood at present.

2.2.3 Environmental Reactivity. Aromatically-bound chlorine found in PCB molecules, deactivates the aromatic ring with respect to electrophilic substitution. In molecules such as PCBs where several chlorine atoms often occur in each ring, the reactivity is very much reduced. This has been generally established for chemical reactivity and it appears that the same pattern exists for biological reactivity, at least under aerobic conditions. Transmission of the deactivating effect between the two rings requires the rings to be in the same plane; substitution in the ortho- (or 2-) positions reduces this coplanarity and affects the environmental properties. Another substituent-related property affecting environmental reactivity is the availability of viscinal aromatic carbon centres for biological attack. Under aerobic conditions, lack of such a structural feature greatly reduces the rate of biodegradation (Neely, 1983). PCBs are not readily degraded in the environment and the higher chlorinated congeners are particularly resistant to degradation. Substituent position also affects other biological properties including toxicity.

PCBs are widely present in the atmosphere; therefore, problems concerning PCBs must be considered on a multi-national or even global level. While the quantities present in the atmosphere are small compared to world production or other environmental compartments (Nisbet and Sarofim, 1972), there is ample evidence to show that the atmosphere is extremely important as a major means of distributing PCBs throughout the environment. Fluxes are important, with the aquatic environment probably providing both the source and sink of the dynamic load in the atmosphere. It is regrettable that there is not much information pertaining to the photodegradation of PCBs under atmospheric conditions (Zabik, 1983). Solution experiments indicate that dechlorination of PCBs to yield lower chlorinated congeners might be the expected result (Hutzinger et al., 1974) but other work (Kotzias et al., 1980) indicates PCBs are unreactive when associated with particulate matter and photo-irradiated. In the vapour phase, the 2,2'-dichloro congener formed hydroxylated transformation products when photo-irradiated in the presence of air.

In water, PCBs do not undergo most hydrolytic reactions, although photolysis is estimated to take place slowly (Bunce et al., 1978) resulting in a stepwise loss of chlorine atoms. This transformation would therefore be expected to give a more biologically reactive substance and speed up degradation. It should also result in decreased bioaccumulation. The contribution of this process to the overall degradation of PCBs in the water compartment, however, is probably small.

Biological degradation of PCBs has been investigated by Liu (1980, 1982) and Tucker et al. (1975). Using an isolated, PCB-degrading micoorganism (*Pseudomonas* sp.), Liu found that the rate of PCB degradation is dependent on chlorine content. Other studies (Furukawa et al., 1979; Masse et al., 1984) have shown similar degradation with other microorganisms, also preselected for their ability to deal with PCBs. The relevance of this to environmental degradation rates is uncertain since it is not possible to relate this to the numbers of these particular strains of bacteria in environmental situations or even in the sewage treatment systems from which they were derived. The value of this type of research is largely to point out the possibility of and to suggest mechanisms which may govern the biodegradation of these compounds; this research also can be used to identify which compounds are more stable.

Martin and Gosselin (1978) did a before and after analysis which showed removal efficiencies of 0 to 90% in a small town sewage treatment system over a five-day period. Another study, by Tabak et al. (1981), failed to detect significant removal with a laboratory system utilizing domestic sewage as an inoculum -- at least for the higher chlorinated mixtures of environmental concern.

The general consensus is that PCBs degrade extremely slowly in the open environment and that their "removal", at least from aquatic ecosystems to which they usually find their way, is dependent mainly on physical processes such as transport out of the system of consideration or covering up via sedimentation. Transportation out of the system simply passes the problem along to the receiving system. Sedimentation as a mode of alleviating the problem is well illustrated by the work of Durham and Oliver (1983) who found that the dated accumulation of PCBs in a core from the Niagara River mouth closely paralleled the production of these compounds, implying both rapid release to the environment and little or no degradation. Based on these studies, it must be concluded that there is no single parameter or even a suite of chemical-specific parameters currently available that would allow the quantitative prediction of the rate of degradation of PCBs in any particular environment -- other than measurements in the system itself.

2.2.4 Modelling of Distribution. An obvious and desirable approach to determining the exposure to a chemical in the environment is to calculate its concentrations in various media (air, water, soil, biota) using physicochemical properties. These are more easily (less costly?) determined than measuring concentrations in the field. In fact, this is often the only option available when evaluations must be done and time and circumstance make other data unavailable. This approach has been attempted using basic parameters such as the several partition constants determined directly or calculated by one of the empirical correlations mentioned (Lassiter et al., 1979; Mackay and Paterson, 1982; Roberts et al., 1978). The models also have added features that, in theory, more closely approximate the real world -- they introduce degradation process rates and the advection of the media themselves. Models have yet to be thoroughly tested in the Canadian scene although the "EXAMS" model (Lassiter et al., 1979) has received some testing in southern U.S. stream situations. A modified version has been described for mirex in Lake Ontario (Halfon, 1982), but it has not been validated or applied to PCBs.

Mackay and Paterson (1982) have indicated from fugacity calculations in their model that currently there is a net diffusion of PCBs in the Great Lakes from the sediment to the water to the air. This is contrary to previous expectations but is not beyond possibility if it is assumed that rain and other PCB inputs to the lakes have actually declined. It is a hypothesis that requires investigation since it implies some success in decreasing PCB concentrations as a result of implementing the existing PCB regulations.

3 ENVIRONMENTAL EXPOSURE TO PCBs IN CANADA

Since the late 1960's, many data have been gathered on the concentrations of PCBs in various constituent parts of the environment (e.g., air, water, soil and biota). In Canada, much of the data development has been carried out by Environment Canada (EC) and the presentation of data in this report follows the regionally organized lines of Environment Canada.

3.1 National Data Banks

Two major attempts at collecting and organizing data related to water quality have been the National Water Quality Data Bank (NAQUADAT) at the Inland Waters Directorate headquarters and the Great Lakes STARFILE Data Bank at the National Water Research Institute. In fact, the two are integrated and except for a time delay in transfering data from STARFILE to NAQUADAT, all data are accessible through the latter. Both of these computer data bases were searched for all references to PCBs subsequent to 1975 -- for all sample types throughout their reporting areas. A summary of the results appears in Table 2.

Data available to May 1985 were mainly those samples collected up to 1983; a small number of data points pertained to 1984 from the Ontario Region (water) and from the Atlantic Region (fish and sediment). For water samples, there was a wide regional disparity in the number of sites examined, the number of samples collected, and the number of those samples in which quantifiable amounts of PCBs were present. Among those with positive results, however, there were no significant differences since the detection levels, depending upon the year of analysis, ranged from 0.009 to 0.020 µg/L. Samples reported as containing PCBs, therefore, were barely quantifiable. The values for the Niagara river are of somewhat greater significance since they mainly pertain to a single station at Niagara-on-the-Lake and represent much more frequent sampling (monthly in 1978 to weekly in 1983) than other sites reporting in NAQUADAT. The mean and 95% confidence intervals for filtered water at this site were 0.020 µg/L and 0.003 µg/L, respectively.

None of the water samples from western Canada appear to contain PCBs at quantifiable levels despite the widespread nature of this pollutant in other parts of the country and its appearance in environmental areas other than those reported in NAQUADAT. Whether or not this is the result of different analytical procedures employed by the various laboratories undertaking these surveys is unknown. The reported

TABLE 2 NAQUADAT PCB DATA (to May, 1985)

Parameter/Region	No. of Sites	No. of Samples	No. of Samples Over Quantification	Mean Conc. of Positives	Collection Year(s)
Water (µg/L)					
Atlantic	37	108	11	0.015	1978-80
Quebec - St. Lawrence	41	280	43	0.040	1978-83
- South Shore	31	67	12	0.038	1980
- Northern	4	14	0	- -	1982-83
Ontario - Niagara	37	413	146	0.021	1974-83
- St. Lawrence	31	118	0	- -	1975-83
- St. Mary's	28	93	0	- -	1973-75
- N.W. Ontario	4	92	32	0.023	1981-84
- James Bay	6	78	25	0.037	1981-83
Western & Northern - Manitoba	17	465	0	- -	1978-83
- Saskatchewan	12	89	0	- -	1978-83
- Alberta	25	815	0	- -	1978-83
- N.W.T.	1	5	0	- -	1980
Pacific & Yukon - Alta./B.C.	9	9	0	- -	1978
Sediment (µg/g)					
Atlantic	15	36	6	0.247	1983-84
Quebec - Northern	4	12	0	- -	1982-83
Ontario - Niagara	6	140	134	0.581	1974-83
- St. Lawrence	44	162	19	0.179	1973-77
Fish (µg/g)					
Atlantic	7	42	10	0.014	1983-84

detection levels, however, are comparable to or better than the samples from the west. An aspect of some concern to the evaluation of national exposure estimates is the paucity of water (and other) data from the Pacific & Yukon Region and from the territories part of the Western & Northern Region.

Sediment data are limited to the three eastern Regions. As with the water, data for Niagara-on-the-Lake dominate the report for the Niagara River. In this case, the mean and 95% confidence limits are 0.581 µg/g and 0.103 µg/g, respectively. Somewhat lower levels were found in the Ontario portion of the St. Lawrence River and in the

Atlantic Region. A few samples from the northern Quebec Region were investigated but no PCBs were found.

A number of fish samples from the Atlantic Region were also reported. The levels in these fish were much lower than those reported for many species found in the Great Lakes.

The managers of the NAQUADAT data base attempt to collect and make available data on PCBs and other substances but they are unable to present a complete picture since much of the data are generated outside the Inland Waters Directorate and do not automatically come to their attention. There is no mandate to search out such data from the literature or from other governmental sources. Some of these other data appear in this report but there is no suggestion that all of it is here; what is here, however, is believed to be representative.

3.2 Atlantic Region

The Canadian Network for Sampling Organic Compounds (CANSOC) made the first attempt at determining atmospheric PCBs in the Atlantic provinces. They produced results for rain over the period 1977-80 for Gander, Newfoundland and Truro, Nova Scotia (Brooksbank, 1983). There were PCBs reported in 70% of the samples from Gander. For Truro, there were less frequent reports (<15% of the samples) of PCBs in rain at or above the quantification level. Subsequent to this, Brun (1982, 1983 and 1984) and Brun and Bailey (1983), reported on studies on wet precipitation* at Kejimkujik National Park in N.S. and at Ellerslie, P.E.I. and on snowpack from Prince Edward Island and nearby parts of New Brunswick and Nova Scotia. In 1984, an additional study at Kouchibouguac National Park in New Brunswick reported somewhat lower values for rain. While the locations are different than those in Brun's studies and a local source has been suggested to account for Ellerslie's values, the differences are unexplained at present. These levels are presented in Table 3.

Some of the more useful data showing PCB trends in the Atlantic region are from seals. Addison et al. (1984) reporting on 1982 grey seal females and pups from Sable Island, N.S., notes that there was little reduction in PCB levels in their blubber from levels reported in a previous study (Addison and Brodie, 1977). Indeed, in the case of pup blubber, the levels were slightly higher than in the earlier study. Data from this study are summarized in Table 4 along with data for other species. PCBs were also found in seal

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* Wet precipitation - rain, sleet, snow, hail.

TABLE 3 ATMOSPHERIC PCB DATA (Atlantic Region)

Location	Sample Type	No. of Samples	No. of Samples Over Quantification	Mean of Positives (µg/L)	Collection Year(s)	Reference
Gander, Nfld.	R	13	9	0.019	1977-80	Brooksbank, 1983
Truro, N.S.	R	27	4	0.021	1977-80	Brooksbank, 1983
Kejimkujik, N.S.	WP	15	4	0.046	1980-81	Brun, 1982
Ellerslie, P.E.I.	WP	15	10	0.195	1980-81	Brun, 1982
P.E.I. (11 sites)	S	11	11	0.0046	1981-82	Brun and Bailey, 1983
N.B. (3 sites)	S	3	3	0.0040	1981-82	Brun and Bailey, 1983
N.S. (3 sites)	S	3	3	0.0019	1981-82	Brun and Bailey, 1983
Kejimkujik, N.S.	WP	12	7	0.070	1982	Brun, 1983
Ellerslie, P.E.I.	WP	12	10	0.210	1982	Brun, 1983
Kejimkujik, N.S.	WP	12	4	0.027	1983	Brun, 1984
Ellerslie, P.E.I.	WP	12	9	0.170 (1)	1983	Brun, 1984
Kouchibouguac, N.B.	R	8	6	0.0011	1984	

Note: (1) does not include one high value of 1.7 µg/L in January

R - rain
WP- Wet precipitation
S - Snowpack

milk and it was suggested that this is a major method of PCB loss for maternal seals (Addison et al., 1984). Direct comparisons between each study cannot be made easily because the mean ages of the seals were quite different (6.8 years in 1976; 14.5 in 1982). The authors concluded, however, that there was little change in the PCB levels in the food sources of these mammals.

Another species for which some "trend" data exists is the Harbour porpoise. Specimens from the Bay of Fundy and other areas on the Atlantic seacoast were studied from 1971 to 1977 (Gaskin et al., 1983). No statistically significant changes in PCB concentrations from blubber or other tissue were noted over this period of time, again implying little change in PCB levels in the food sources of these animals. The lack of

TABLE 4 PCBs IN BIOTA FROM THE ATLANTIC REGION

Species	Location	Sample Type	No. of Samples	PCB Concentration (μg/g)	Collection Year(s)	Reference
Grey seal	Sable Island	mothers (whole body)	8	11.7	1976	Addison et al., 1984
		pups (whole body)	8	1.9	1976	Addison et al., 1984
		mothers (whole body)	6	15.7	1982	Addison et al., 1984
		pups (whole body)	6	5.0	1982	Addison et al., 1984
Harbour porpoise	Fundy	male (blubber)	62	78.7	1971-77	Gaskin et al., 1983
		female (blubber)	40	45.0	1971-77	Gaskin et al., 1983
	Rhode Island/ Newfoundland	all blubber	5	55.0	1971-73	Gaskin et al., 1983
Greater shearwater	Fundy	male (fat)	8	35.0	1974	Gaskin et al., 1978
		female (fat)	10	72.9	1974	Gaskin et al., 1978
Sooty shearwater	Fundy	male (fat)	4	17.2	1974	Gaskin et al., 1978
		female (fat)	2	19.7	1974	Gaskin et al., 1978

regional differences in porpoise blubber levels from Rhode Island, U.S.A. to Newfoundland suggests a common mechanism for PCB absorption in these mammals.

Two other species from the area that have been examined are from the puffin family -- the Greater shearwater and the Sooty shearwater. Specimens collected in 1974 (Gaskin et al., 1978) from the Bay of Fundy area showed substantial levels of PCBs in fatty tissue, although the variability was high. Data were also presented for other organs from these species but comparable conclusions were drawn in each case despite the lower PCB concentrations found in organ tissue.

Fish is the primary food source for these birds. There is little new data on PCB concentrations in these animals; however, Travers and Wilson (1977) describe the state of affairs prior to 1977 (much of this information appears in the Task Force (1976) Report).

3.3 Quebec Region

Data for the Quebec Region were gathered from areas of major human population in the St. Lawrence River and Gulf areas. Hopefully more data on the northern hinterland will be obtained in the future since this represents such a large fraction of Quebec's environment which also requires protection.

Direct measurement of PCBs in atmospheric precipitation exists only in the CANSOC network. Data for three stations (St. Hubert, Maniwaki, and Chibougamau) indicated that PCBs were present in only 50% of samples, and in those samples, levels were comparable to those observed elsewhere. In addition, Ware and Addison (1973) found that there was a significant correlation between the amount of rain falling 10 to 20 days before sampling and the concentrations of PCBs in plankton from the southern part of the Gulf of St. Lawrence. This suggests that the PCBs originated from the atmosphere.

Data gathered for the Quebec region were varied, and comparisons within the region were difficult because data originated from different sample types and times. Selected data are presented in Table 5.

A comparison can be made between the situation up stream and down stream of Montreal on the St. Lawrence River. In edible tissues from a variety of fish, concentrations were greater by a factor of three (approximately) for upstream samples -- a relationship that existed in both 1976 and 1984 (Sloterdijk, personnal communication). This apparently results from pollution up stream of Lake St. Francis from sources on the Grasse River near Massena, N.Y.; an ALCOA (aluminium) plant and a General Motors foundry in the area are indicated sources (IJC, 1984). In addition to the contamination in the Lake St. Francis area, high levels of PCBs were found in fish collected in 1984 from Lake St. Louis (south shore downstream from the Beauharnois canal). Values from these samples contributed greatly to the 1984 upstream mean indicated in Table 5. The suspended load may also contribute to the upstream/downstream difference. Up stream, the suspended material is found at 4 to 10 mg/L while down stream of Lake St. Peter to Quebec City, levels ranged from 30 to 50 mg/L (Couillard, 1982a). The dilution of particulate-sorbed PCBs would probably find its reflection in lower concentrations in the biota that utilize this material as part of their food web.

It is difficult to describe temporal trends for the region since, as mentioned, the samples are not easy to compare. Current research and data from Goulet and Laliberté (1982), however, provide some indication of trends. Concentrations in walleye, with and without those for pike, were at comparable levels in 1976 and 1978; white suckers, together with information on the brown bullhead, show a 50% decrease over this time period. A longer period was available for the perch together with pumpkinseeds, which indicated a 77% reduction between 1976 and 1984. None of these comparisons are statistically significant but at least the data do not indicate an increase!

A considerable study on the St. Lawrence River was undertaken by federal and provincial agencies. Much of the data relating to PCBs from that study were reviewed by

TABLE 5 PCB DATA (Quebec Region)

Species	Location	Sample Type	No. of Samples	PCB Concen-tration (µg/g)	Collection Year	Reference
St. Lawrence River						
Walleye and Northern Pike	upstream Montreal	flesh	?	2.7	1976	
	downstream Montreal	flesh	?	1.0	1976	
Walleye	downstream tributary	whole	28	0.86	1978	Goulet and Laliberté, 1982
White Sucker and Brown Bullhead	upstream Montreal	flesh	?	2.5	1976	
	downstream Montreal	flesh	?	1.1	1976	
White Sucker	downstream tributary	whole	39	0.55	1978	Goulet and Laliberté, 1982
Yellow Perch and Pumpkinseed	upstream Montreal	flesh	?	0.9	1976	
	downstream Montreal	flesh	?	0.6	1976	
Perch	upstream Montreal	flesh	?	0.35	1984	
	downstream Montreal	flesh	?	0.14	1984	
Spottail Shiner	upstream Montreal	flesh	?	0.79	1984	
	downstream Montreal	flesh	?	0.16	1984	
Other fish	downstream tributary	whole	7	0.71	1978	Couillard, 1982b
Clams	downstream tributary	whole	37	0.016	1978	Couillard, 1982b
Great Blue Heron	inland, river and estuary	eggs	47	7.9	1979	Laporte, 1982
Churchill Falls/River						
Burbot	above power plant	liver	22	0.48	1977	Musial et al., 1979
	below power plant	liver	9	17.6	1977	Musial et al., 1979
Lake Trout	above power plant	fat	8	0.50	1977	Musial et al., 1979
Gulf/Estuary St. Lawrence						
Sediment	Baie des Chaleurs	whole	45	0.0049	1979	Beak, 1980
Bivalves	Baie des Chaleurs	whole	17	0.075	1979	Beak, 1980
Lobster	Baie des Chaleurs	whole	23	0.0122	1979	Beak, 1980
"Fish"	Baie des Chaleurs	whole	33	0.0196	1979	Beak, 1980
Plankton	north of P.E.I.	>73 µm, whole	9	1.39	1972	Ware and Addison, 1973
Harp seals	Gulf	blubber	11	7.7	1971	Addison et al. 1984
	Gulf	blubber	22	4.0	1982	Addison et al., 1984

Couillard (1982b), some of which also appear in Table 5. The data for Atlantic herring indicate an input of PCBs to the estuary from the Saguenay River and preferential transport along the north shore, although not exclusively so (Couillard, 1982b). A report by Khalil and Arnac (1975) also had qualitatively similar results. Generally, however, concentrations in species of the Gulf and estuary areas are lower than those up stream of Quebec City.

Great Blue herons collected from the St. Lawrence River and Gulf area in 1979 showed marginal differences among weighted averages for the river, estuary and gulf (Baie des Chaleurs) although those from "inland" areas were lower (Laporte, 1982). A small number of Razorbill eggs were also examined for PCBs (Chapdelaine and LaPorte, 1982). These showed a considerable difference between gulf and estuary concentrations although it was not deemed to be responsible for the reduced populations in the estuary. Pearce et al. (1979) studied a variety of seabird eggs from this region and the Atlantic provinces and found that there were few significant differences in geographical and temporal levels between 1972 and 1978.

Harp seals have also been investigated in the Gulf area. Results for samples collected in 1971 and 1982 show no significant changes although the means for PCB concentrations were 7.4 µg/g and 4.0 µg/g (wet weight, blubber) for each year respectively, suggesting an improvement (Addisson et al., 1984).

Some data are available for the Quebec Region pertaining to inland Quebec/-Labrador and to the Baie des Chaleurs. These data are not numerous but do serve to demonstrate the widespread nature of the PCB problem. A source of PCBs to the Churchill River appears to be from the power site, at least according to burbot data (Musial et al., 1979). Lake trout, another top predator fish, do not have a complete enough data set to draw the same conclusion. Levels up stream of this source are much reduced when compared to data for the same species in the Great Lakes.

A 1980 study under the auspices of Environment Canada's "Environmental Contaminants Contract Fund" noted substantial concentrations of PCBs in sediments, bivalves, lobsters and "fish" collected in Baie des Chaleurs in 1979 (Beak, 1980). This was in contrast with statements made about the absence of PCBs in the Gulf (of which the bay is a part) noted previously and in the Task Force Report. Whichever is correct, it should be pointed out again that these levels, while undesireable, are not as elevated as those in other species from the Great Lakes.

In an examination of possible sources for the PCBs observed in species from this region, effluents from a selected number of industrial operations were investigated during 1977-79 (Goulet and Laliberté, 1982). Significant releases were observed for a paper recycling plant in Candiac (130 g/d); an aluminium producer in Arvida (28 g/d); and a paint company in Montreal (<5 g/d). Together, these three industries represent a release of about 160 g/d or an annual loss of 60 kg. These losses, however, are not appropriate for their industries as a whole since other plants in the same industry sectors did not appear to have such releases. Two other sectors, the chemical industry and tanneries, showed no sign of PCB losses. In any event, it is probable that these losses would be small when compared with atmospheric deposition and PCBs being exported from the Great Lakes through the St. Lawrence River.

3.4 Ontario Region

In Canada, most of the PCB data documented is for the Ontario Region. The Great Lakes, which are located in this region are collectively the largest body of freshwater in the world and their large areas, together with their long retention times (up to 175 years) make them vulnerable to contamination with persistent organics. PCBs have been reported for the Great Lakes system for almost two decades now but, as with other areas of Canada, the data are not always of the sort that allow extrapolation or interpolation since this requires comparability both in sample type and consistency over time. There are, however, several data sets which permit such trend evaluation; the Herring Gull egg monitoring program is the oldest of these. Summary data from this and other programs are presented in Table 6 although, for brevity, only the more recent years are given with the exception of the years which correspond to the commencement of the several programs.

The lake trout and the rainbow smelt data were provided by Fisheries and Oceans (IJC, 1983; D.M. Whittle, pers. com.); the spottail shiner data by the Ontario Ministry of the Environment (IJC, 1983; W. Scheider and K. Suns, pers. com.); and the Herring Gull egg data by the Canadian Wildlife Service of Environment Canada (IJC, 1983; D.V. Weseloh, pers. com.). Many of these data are documented each year in reports to the International Joint Commission which has the responsibility of administering the Great Lakes Water Quality Agreement (IJC, 1983). Response to the requirements of this Agreement have provided much of the impetus for the development of the trend data that is available for the Ontario Region.

From the Herring Gull egg data it can be seen that residue levels in the Great Lakes have improved. There is a fluctuation in the trends of levels for each lake and for all lakes, there appears to have been some increase in PCB levels during 1981-82.

TABLE 6 TREND DATA FOR PCBs IN SELECTED GREAT LAKES BIOTA

Year	Concentration (μg/g)	No.	Rainbow Smelt Concentration (μg/g)	No.	Spottail Shiner Concentration (μg/g)	No.	Herring Gull Eggs** Concentration (μg/g)	No.
Lake Superior	(Lake Trout)*							
1974	-	-	-	-	-	-	62.	19
1977	1.7	4	-	-	-	-	55.	20
1979	0.68	?	-	-	0.03	44	59.	20
1980	0.85	50	-	-	-	-	26.	20
1981	0.43	58	0.11	12	-	-	34.	20
1982	0.50	31	0.16	12	-	-	35.	20
1983	0.71	101	0.23	22	0.07	62	21.***	11
1984	-	-	-	-	0.03	28	17.	20
Lake Huron	(Lake Trout)*							
1974	-	-	-	-	-	-	71.	20
1977	2.0	4	-	-	-	-	70.	20
1980	2.4	50	0.11	36	0.05	42	20.	20
1981	2.3	50	0.13	36	-	-	25.	20
1982	2.4	79	0.29	24	-	-	34.	20
1983	1.2	50	0.18	24	-	-	19.	22
1984	-	-	-	-	-	-	20.	20
Lake Erie	(Walleye)							
1974	-	-	-	-	-	-	72.	20
1975	-	-	-	-	0.84	5	62.	20
1977	1.6	9	0.18	78	0.30	24	69.	20
1980	1.4	40	0.26	42	0.21	21	46.	19
1981	1.2	44	0.23	36	0.24	17	56.	20
1982	1.6	30	0.30	34	0.12	33	59.	11
1983	1.5	50	0.32	29	0.14	25	37.	21
1984	-	-	-	-	0.23	27	46.	20
Lake Ontario	(Lake Trout)*							
1974	-	-	-	-	-	-	152.	19
1975	-	-	-	-	0.69	5	143.	20
1977	8.0	3	1.5	38	1.49	15	102.	20
1980	4.8	82	1.1	33	0.28	34	56.	19
1981	3.7	83	0.90	47	0.49	19	79.	20
1982	5.9	36	1.7	44	0.24	32	62.	19
1983	6.4	46	1.5	36	0.33	49	43.	22
1984	5.9	47	1.7	24	0.18	40	51.	20

Notes:

* Lake trout from Lake Ontario are from the 4+ year class; 1977 data from the Upper Lakes are not from the same agency although they are comparable

** eggs are collected from two colonies in each lake each year

*** eggs were collected from only one site on the lake this year

Fortunately, however, levels seem to have returned to their general downward trend in 1983.

Lake Ontario is the most polluted of the Great Lakes with respect to PCBs. Data from Lake Ontario lake trout demonstrated similar trends to the data for Herring Gull eggs. Since 1982, PCB levels in lake trout have been increasing; however, 1984 data suggest that levels in lake trout may again be dropping. The data for the rainbow smelt are ambiguous as to trend; those for the spottail shiner can only be claimed to "indicate" a downward trend. No explanation of the 1981-82 reversal is offered although it is easy enough to speculate about either a new source to the lake or a "leveling-off" or re-equilibrating of the system following initial success with PCB controls in the late 1970's. The 1984 lake trout data offer some hope that the upward trend for this species has also stopped, but this remains to be established in future studies.

Lake trout data for the other lakes are ambiguous with few trends being apparent. Residue levels for walleye in Lake Erie are not strictly comparable with those for lake trout in the rest of the lakes. Lake trout are the top predators in the Great Lakes except in Lake Erie where the top predator is walleye. Generally, the upper lakes have lower contaminant levels, reflecting, it is presumed, the lower density of population, fewer industrial sources, and possibly the atmospheric inputs to waterbodies with a higher volume to area ratio.

Trends in rainbow smelt data are not clear as yet, although the relative differences between lakes are similar to those observed with lake trout. The spottail shiner program is a promising one because this species is localised in its habitat and the program utilizes only "young-of-the-year" specimens (10 individuals composited per sample). A consequence of this is that the data reflect current, local conditions but comparisons on a lake-wide basis are risky unless the data pertain to many locations in each basin or to the same sites each year. This has not always been the case. PCB residues in spottail shiners in the lower lakes appear to show a downward trend; data for the upper lakes are not available for a comparable evaluation.

The Ontario Ministry of Environment undertakes extensive monitoring of various fish species and the data are used to produce an annual "Guide to Eating Ontario Sport Fish". Some of the data for 1983 which exceeded 1.0 μg/g (arbitrary value) are presented in Table 7. It is apparent that most of the higher levels occur in the lower Great Lakes system and that only two of the inland lakes reached this level despite being sampled just as intensively. There are, however, many inland lakes in which one or more species has measureable levels of PCBs, the significance of which is presently unknown.

TABLE 7 ADDITIONAL PCB DATA FOR THE GREAT LAKES

Location	Sample Type	No. of Samples	PCB Concentration	Collection Year(s)	Reference
L. Ontario (open)	water (unfiltered)	14	0.00078 µg/L	1983	McCrea et al., 1985
	sediment	216	0.058 µg/g	1968	Thomas and Frank, 1983
L. Ontario Cdn. trib. (ex. Niagara)	susp. sediment	34	0.16 µg/g	1974	Frank et al., 1981
	" "	4	0.035 "	1975	Frank et al., 1981
	" "	13	0.013 "	1977	Frank et al., 1981
Niagara River (lower)	Water (filtered)	1	0.011 µg/L	1981	Wagner et al., 1984
	" "	75	0.020 "	1980-81	Wagner et al., 1984
	" "	7	0.00075 "	1981	Wagner et al., 1984
	susp. sediment	8	0.58 µg/g	1979	Wagner et al., 1984
	" "	41	0.96 "	1980	Wagner et al., 1984
	" "	6	0.49 "	1981	Wagner et al., 1984
L. Erie (open)	sediment	259	0.095 "	1971	Thomas and Frank, 1983
L. Erie Cdn. trib (ex. Detroit)	susp. sediment	11	0.030 "	1975	Frank et al., 1981
	" "	8	0.018 "	1977	Frank et al., 1981
L. St. Clair	sediment	55	0.019 "	1970	Thomas and Frank, 1983
	"	55	0.010 "	1974	Thomas and Frank, 1983
L. Huron (open)	sediment	174	0.013 "	1969	Thomas and Frank, 1983
Georgian Bay/N. Chan.	"	170	0.011 "	1973	Thomas and Frank, 1983
L. Huron/Georgian Bay/ N. Chan. trib. (ex. St. Mary's)	susp. sediment	9	0.085 "	1974	Frank et al., 1981
	" "	8	0.022 "	1975	Frank et al., 1981
	" "	15	0.025 "	1977	Frank et al., 1981
L. Superior (open)	sediment	405	0.0033 "	1973	Thomas and Frank, 1983
L. Superior Cdn. trib.	susp. sed.	3	0.023 "	1975	Frank et al., 1981
Ontario Ministry of Environment (unpublished data)					
L. Ontario/Credit R.	Chinook salmon	20	4.0 µg/g	1983	
	Rainbow trout	20	2.6 "	1983	
	Coho Salmon	20	2.6 "	1983	
L. Ontario/Ganaraska R.	Rainbow trout	20	1.6 "	1983	
	Brown trout	19	3.1 "	1983	
L. Erie/Central Basin	Channel catfish	20	1.8 "	1983	
L. Erie/Western Basin	" "	20	2.8 "	1983	
L. St. Clair	" "	30	1.5 "	1983	
St. Clair R./Sarnia	Gizzard shad	6	1.6 "	1983	
	Carp	2	1.4 "	1983	
L. Huron/South	"	20	2.8 "	1983	
L. Huron/Bruce Penn.	"	20	2.0 "	1983	
	Channel catfish	20	1.4 "	1983	
L. Simcoe	Whitefish	15	1.0 "	1983	
Rice Lake	Carp	20	1.9 "	1983	

In the Ontario Ministry of Environment presentation at a Health and Welfare Canada workshop (November, 1985) the observed general downward trend for PCB levels in fish previously mentioned was also reported for rainbow trout from the Ganaraska R., (L. Ontario); for coho salmon from the Credit R. (L. Ontario); and for splake from Cape Rice (L. Huron).

Sediment samples from the Great Lakes are good time-integrators of lake conditions. Thomas and Frank (1983) have recently summarized the data on PCBs in surficial sediments; lake means are presented in Table 7. It is apparent that the lower lakes have higher PCB concentrations than the upper lakes (as do the fish, generally). Atmospheric deposition might be expected to result in comparable levels, but the higher human population density around the lower lakes is probably responsible for the higher concentrations there through other input mechanisms.

Data are also more abundant in this region on the state of PCBs in the atmosphere. A summary of available Canadian data appears in Table 8. Most of the atmospheric PCB concentration data were obtained from studies done by the Ontario Ministry of the Environment (Singer et al., 1983). This ministry altered its analytical procedures in 1980 and one result was that the levels of PCBs in air were one to two orders of magnitude lower than previously recorded. Whether or not the PCB levels in air actually decreased or this was a function of the change in analytical methods is still unknown. Data in Table 8 indicate that there is little significant difference between air residue levels in the northern and southern parts of the province. For the Great Lakes region, Eisenreich et al. (1980, 1981) estimated that air concentrations of approximately 1 ng/m^3 were appropriate and that this would result in the dry deposition of 1.8 to 7.8 tonnes of PCBs to each lake each year; Rice et al. (1982) determined a value of 3.1 ng/m^3 for nearby Lake Michigan.

There are qualitative indications that the residue levels in precipitation have decreased for Lake Superior and if so, by inference, to the rest of the Great Lakes system and possibly even to the rest of Canada. Calculations by Eisenreich et al. (1980, 1981) using a concentration value of 0.030 μg/L indicated that wetfall alone could account for an input of 0.5 to 2 tonnes of PCBs per lake per year. Current levels of PCBs in rain (mean 0.003 μg/L) would reduce those estimates by a factor of ten. The lower concentration was determined from "open-lake" samples of > 90% of 1983 rain; the previous value (0.030 μg/L) was for comparatively fewer rain events representing approximately 25% of the annual rainfall during 1976-1977.

TABLE 8 ATMOSPHERIC PCB DATA (Ontario)

Sample Area	No. of Sites	No. of Samples	PCB Concentration	Collection Year(s)	Reference
Air			(ng/m^3)		
L. Ontario	3	34	2.6	1975	Singer et al., 1983
	14	352	6.2	1979	Singer et al., 1983
	12	167	0.15*	1980	Singer et al., 1983
Windsor	1	27	5.6	1979	Singer et al., 1983
	1	13	0.06*	1980	Singer et al., 1983
Other S. Ontario	6	146	6.7	1979	Singer et al., 1983
	8	111	0.23*	1980	Singer et al., 1983
L. Superior	2	53	4.8	1979	Singer et al., 1983
	2	26	0.05*	1980	Singer et al., 1983
Other N. Ontario	2	50	5.6	1979	Singer et al., 1983
	2	27	0.02*	1980	Singer et al., 1983
Rain			(μg/L)		
L. Ontario	2	16	0.032	1976	Strachan and Huneault, 1979
	2	10	0.022	1977	Strachan et al., 1980
Windsor	6	?	0.094 (estimate)	1975-1977	Sanderson and Lavalle, 1979
	4	31	0.022	1984	GLI, 1985
L. Superior	2	14	0.026	1976	Strachan and Huneault, 1979
	1	4	0.014	1977	Strachan et al., 1980
	2	24	0.0063	1983	Strachan, 1985
	2	30	0.0029	1984	
Other Ontario	3	20	0.010	1976	Strachan and Huneault, 1979
	4	17	0.022	1977	Strachan et al., 1980
Snow					
Southern Ontario	4	4	0.020	1977	Strachan and Huneault, 1979
	13	50	0.0012**	1978	Beak, 1979
Northern Ontario	13	13	0.032	1977	Strachan and Huneault, 1979
	15	54	0.0005**	1978	Beak, 1979

Notes: * a change in analytical procedures is the reason for the dramatic "improvement" in the 1980 samples

** the data quality is uncertain

Quantities that are input to Lake Ontario from the Niagara River can be estimated from the annual flow characteristics (2×10^{11} m^3 water, 1.8×10^9 kg suspended sediment (Kuntz and Warry, 1983)) and the PCB concentrations listed in Table 7. These, and similar estimates, result in estimates for total input via water plus suspended solids in excess of 2 tonnes, a substantial portion of which is added along the course of the river itself. Whittle and Fitzsimons (1983) investigated residue levels in several species occurring in both the eastern basin of Lake Erie (the input to the Niagara) and the western portion of Lake Ontario (the output). These all form part of the food web of predator fish such as trout and salmon and all showed substantial residue level increases in going from Lake Erie to Lake Ontario.

In an examination of the sediments, Kauss (1983) found little significant difference between the upper and lower portions of the Niagara River although there were much higher levels in suspended solids from the upper Niagara, mainly on the American side. Other comparisons have been summarized in the report of the Niagara River Toxics Committee (Wagner et al., 1984); these include water (0.00026 µg/L for Lake Erie versus 0.00075 µg/L for Lake Ontario), sediments (0.086 µg/g vs 0.61 µg/g), plankton (0.16 µg/g vs 0.28 µg/g), spottail shiners (0.060 µg/g vs 0.26 µg/g), rainbow smelt (0.21 µg/g vs 0.86 µg/g) and smallmouth bass (0.85 µg/g vs 3.1 µg/g). Similar data for the Detroit and St. Mary's Rivers are not readily available although the Upper Great Lakes Connecting Channels Committee (a joint Canada-U.S. study group) are in the process of compiling these data. Annual flows of 1.6×10^{11} m^3 carrying a suspended solids load of 3.2×10^9 kg contaminated with 0.01 to 0.1 µg/g of PCBs would suggest that the Detroit River contributes 30 to 300 kg of PCBs per annum to Lake Erie via the suspended load (Platford et al., 1985).

Another mode of entry to the lakes is the input from tributaries. The Pollution from Land Use Activities Reference Group of the I.J.C. are responsible for gathering data which allows the estimates for this mode to be made. The more recent data are summarized in Frank et al. (1981) and are further condensed in Table 7. As with most rivers in Canada, there are considerable differences in seasonal flows making estimates of annual loading difficult since sampling and analyses are seldom done frequently enough. Frank et al. (1981) estimated the total annual Canadian loadings of PCBs to Lake Ontario (excluding the Niagara River) to be 4.7 kg; to Lake Erie (excluding the Detroit River) 5.9 kg; to Lake Huron/Georgian Bay/North Channel (excluding St. Mary's River) to be 5.6 kg; and to Lake Superior 7.8 kg. Even allowing for the larger American population which might contribute twice this amount based upon population, these loadings are small when compared to PCB inputs via the atmosphere and via the connecting channels.

Urban runoff has also been mentioned as a potential loading source for the Great Lakes waters, which have water from many towns and cities draining either directly or via tributaries to them. Marselek and Schroeter (1984) examined runoff from 119 Ontario communities in the basin area and concluded that annual estimates for loading to each lake in this manner ranged from 2.0 to 66.5 kg/yr; the total to all lakes was 121 kg/yr. These amounts, as with the tributary loads, are small in comparison with the previously mentioned sources and may have been partly derived from the atmosphere.

It should be noted that there have also been investigations done in the Ontario Region outside the Great Lakes basin. The major rivers flowing into Hudson Bay and James Bay were examined at their mouths during 1981 (McCrea et al., 1984). The data in Table 9 show a lower level of biota contamination than is generally found in the Great Lakes part of the Ontario Region but the significant aspect is that PCBs are found at all in this area. The implication must be for a widespread as opposed to local contamination source and the atmosphere is an obvious candidate.

In the data from this area, there is a discrepancy between water values obtained during the special 1981 study and those reported in NAQUADAT (see James Bay in Table 2). The same laboratory analysed both sets of samples. The lower, 1981 values pertain to an experimental, large volume (200-L) sampler which utilized centrifuged water (and gave rise to the suspended solid data also shown in Table 9). Results for 1981-83 are for samples of 1 to 2 litres treated according to standard procedures. It is not clear if the samples processed in this way were filtered or not. It may also be that the high flow period of the special study may give anomolously low values.

The Ottawa River has also been examined near Ottawa - in studies during 1977 and 1980 (Merriman and McCrea, 1985). In 1980, PCBs over the quantification limit of 0.01 µg/L were found mainly in spring samples; they were also observed at trace levels throughout the year. The mean water concentration of a comparable transect of the river (Templeton-Quebec/Gloucester-Ontario) indicated roughly half the concentration in 1980 compared to that of 1977.

In northwestern Ontario, levels similar to those reported for the Ottawa River (Water Quality Branch - Ontario, personal communication) have been found. The frequency of observation was somewhat higher.

TABLE 9 PCB DATA FROM OUTSIDE THE GREAT LAKES (Ontario)

Location	Sample Type	No. of Samples	No. over Quantification	Mean of Positives	Collection Year(s)
Northeastern Ontario (McCrea et al., 1984)					
Moose River	water	1	1	0.00025 µg/L	1981
	"	21	14	0.025 "	1981-83
	suspended sediment	1	1	0.00129 "	1981
	sediment	1	1	0.01 µg/g	1981
	white sucker	5	5	0.074 "	1981
	northern pike	5	5	0.036 "	1981
Albany River	water	1	1	0.00021 µg/L	1981
	"	9	5	0.009 "	1981-83
	suspended sediment	1	1	0.00090 "	1981
	sediment	1	0	-- µg/g	1981
	white sucker	5	5	0.074 "	1981
	northern pike	5	5	0.056 "	1981
Attawapiskat River	water	1	1	0.00024 µg/L	1981
	"	11	4	0.018 "	1981-83
	suspended sediment	1	1	0.00023 "	1981
	sediment	1	0	-- "	1981
	white sucker	3	1	0.030 µg/g	1981
	northern pike	5	3	0.013 "	1981
Winisk River	water	1	1	0.00043 µg/L	1981
	"	15	4	0.015 "	1981-83
	suspended sediment	1	1	0.00117 "	1981
	sediment	1	0	--	1981
	white sucker	3	1	0.010 µg/g	1981
	northern pike	5	5	0.012 "	1981
Severn River	water	1	1	0.00024 µg/L	1981
	"	12	8	0.021 "	1981-83
	suspended sediment	1	1	0.00157 "	1981
	sediment	1	1	0.01 µg/g	1981
	white sucker	5	0	--	1981
	northern pike	5	2	0.015 µg/g	1981
Ottawa River near Ottawa (Merriman and McCrea, 1985)					
Templeton/Gloucester	water	7	3	0.067 µg/L	1977
	"	12	2	0.020 "	1980
Other incl. tributaries	"	36	6	0.019 "	1980
Northwestern Ontario (Water Quality Branch - Ontario, unpubl. results, 1985)					
Rainy River (Rainy Lake)	water	20	8	0.021 µg/L	1981-83
Rainy River (Lake-of-the Woods)	"	23	8	0.026 "	1981-83
Winnipeg R. (Whitedog Falls)	"	24	9	0.021 "	1981-83
English R. (Caribou Falls)	"	23	10	0.019 "	1981-83

3.5 Western and Northern Region

Outside the data mentioned previously from NAQUADAT (Table 2), there are only a few new studies on PCB exposure in the prairie provinces. Fish in the south and central parts of the province of Alberta were recently investigated together with sediments from the North Saskatchewan River (Chovelon et al., 1984). Data pertain to specimens collected between 1980 and 1982 and are summarized in Table 10.

TABLE 10 PCB DATA (South/Central Alberta) (Chovelon et al., 1984)

Species	Location	Sample Type	No. of Samples	Concentration (µg/g)	Collection Year(s)
Fish	N. Sask. River	muscle	?	0.01 to 1.7	1980-82
	Red Deer River	"	?	0.01 to 0.72	1980-82
White Sucker	N. Sask. River	intestinal fat	40	3.66	1980-82
	Battle River	" "	20	0.64	1980-82
	Beaver River	" "	16	0.07	1980-82
	Lake Wabamun	" "	17	ND*	1980-82
Walleye	N. Sask. River	" "	65	14.2	1980-82
	Beaver River	" "	21	1.02	1980-82
	Bow River	" "	17	7.00	1980-82
	St. Mary's River	" "	23	5.01	1980-82
Sediment	N. Sask. River	surficial/dry	160	<0.01	1980-82

* ND - not detected

It is difficult to relate the fat/muscle levels from Table 10 to those in whole fish examined elsewhere. On the assumption that intestinal fat is comparable to other fatty tissue, the levels would appear, at least for some species/locations, to be comparable to those of the Great Lakes. The highest levels were found in the North Saskatchewan River although the variability was high and more specific source assignments could not be made. Sediment levels were all below quantification which was 0.01 µg/g (dry weight). This would seem to be anomalous if results from the fish and sediments are considered. Generally there appears to be no identifiable discharge of PCBs to the North Saskatchewan River.

Data from the CANSOC network were collected from three stations in the region (Fort McMurray, Edson, and Wynyard). From a total of 31 samples collected over the 1977-80 period, eight showed PCBs at a mean value of 0.019 µg/L. An examination of

PCBs and other organochlorine substances in the rainfall from Cree Lake (northern Saskatchewan) in 1984 showed a volume-weighted mean PCB concentration of 0.0011 µg/L. This level, and possible downward trend, are in agreement with observations for Lake Superior.

The northern and arctic areas of this region have more new data available than do the western or prairie areas. Selected data which were provided or found in the literature are summarized in Table 11. They are drawn from a review by Muir (1985) of contaminant levels in mammals and fish from this area and from an examination of polar bear livers from the area by Norstrom (personal communication).

TABLE 11 ARCTIC PCB DATA

Species	Location	Sample Type	No. of Samples	Concentration (µg/g)	Collection Year(s)	Reference
Polar Bear	Eastern Arctic	liver	?	9.9	1984	
	Western Arctic	liver	?	12.4	1984	
	Hudson Bay	liver	?	17.4	1984	
Harp seal	Eastern Arctic	blubber	42	1.95	1976-78	Ronald et al., 1984
Ringed seal	Central Arctic	blubber	50	0.85	1984	Muir, 1985
Beluga whale	Eastern Arctic	blubber	11	1.23	1983	Muir, 1985
	Hudson Bay	blubber	9	2.57	1983	Muir, 1985
Arctic char	all areas	muscle	16(of 43)	0.013	1984	Muir, 1985

As with other regions, these types of data seldom permit easy comparison for indication of geographical or temporal trends. There is, however, one data set which permits speculation. The polar bear, which is the top predator in the area, has higher PCB levels in 1984 samples from the eastern Arctic than those from the western Arctic; those from the Hudson Bay area, however, are even higher. This particular evaluation is made from graph analysis and since the statistical values are not readily determined, confidence in the previous statement is uncertain. Norstrom (personal communication), in attempting to draw comparisons with 1971-72 data from Bowes and Jonkel (1975) (Table 1), concluded that there had been an approximate three-fold increase in the levels of comparable samples. Temporal comparisons are not readily made for seals and whales although there appeared to be no major increase between the early 1970's (Table 1) and the present values for ringed seals.

Arctic data for fish are sparse; however, data for Arctic char, a species closely related to the lake trout of the Great Lakes, indicate that levels of PCB contamination in the Arctic are much lower than those found elsewhere in Canada. Generally, the concentrations found in this species during 1984 were below the quantifiable level of 0.001 µg/g (Muir, 1985). Higher levels were found only at Ellice River, Jayco Lake and at Cambridge Bay where the means were 0.013 µg/g (the maximum concentration was 0.035 µg/g).

A recent investigation at Distant Early Warning (DEWLine) sites identified substantial quantities of PCBs in electrical equipment located there (Holtz and Sharpe, 1984). Five out of 22 abandoned sites were examined; there are a total of 42 such locations, all similarly equipped. Of those examined in 1984, an average of 305 L (460 kg) were estimated to be present making a collective total at all DEWLine sites of 19.2 tonnes. This is a substantial quantity, all of it in remote locations near a latitude of 70° N. The study showed that some of the PCBs were escaping at four out of the five abandoned sites.

3.6 Pacific and Yukon Region

In 1983, a major review of the PCB situation in British Columbia was undertaken by Environment Canada (Garrett, 1983). Extensive data were presented for water, sediment, aquatic invertebrates, fish, birds and various discharges mainly in the lower Fraser River/Estuary area. Other data related to "spills" were given for sediment, surface water, soil and groundwater. Summaries of this data are presented in Table 12. The rain data come from the CANSOC network but were not available to be added to this review.

Data for the largescale sucker allow some comparisons to be made for the Fraser River. The Main Arm and the stretch of river between Hope and Chilliwack both dropped from similar levels in 1973 (0.16 and 0.14 µg/g in muscle) to less than detectable in 1980. The levels for the North Arm of the Fraser River appear to have risen over the same period, from 0.075 to 0.15 µg/g. Levels in northern squawfish during the same time period decreased from 0.42, 0.055 and 0.72 µg/g for the Main Arm, Hope-Chilliwack and North Arm portions of the river, respectively, to less than detectable except for the North Arm which dropped to 0.18 µg/g. These comparisons, which generally show improvement in the system, are not statistically significant due to the wide range of

TABLE 12 PCB DATA (British Columbia) (Garrett, 1983)

Species	Location	Sample Type	No. of Sites	No. of Samples	No. of Samples Over Quantification	Mean of Positive Measurements	Collection Year(s)
						(μg/L)	
Rainwater***	Vancouver	wetfall	1	20	3	0.011	1977-80
Surf. water	various	unknown	14	38	15	0.80	1976-80
						(μg/g)	
Sediment	Fraser est.	surface/dry	50	109	70	0.13	1976-79
	Van. coast/harb.	"	34	88	85	1.32	1976-81
	Burrard Inlet	"	28	59	50	0.98	1976-81
	Howe Sound	"	10	46	32	0.32	1978-81
	N. coastal	"	20	78	69	0.14*	1977-81
	McLeese Lake	"	16	35	17	0.12**	1976-78
	Vanc. Is. coast	"	18	292	263	0.31	1977-81
Mussels	Fraser	wet/composite	3	?	3	0.14	1978-80
	Prince Rupert	"	4	?	4	0.17	1977
	Alice Arm	"	6	?	6	0.24	1980-81
Flounder	Van./Fraser	muscle	5	26	25	0.21	1978-80
	Prince Rupert	whole/ composite	4	?	1	0.017	1977
Largescale sucker	Lower Fraser	muscle	20	46	36	0.15	1973, 1980
	Fraser to Hope	"	7	64	25	0.27	1973, 1980
Rainbow trout	Lower Fraser	"	<17	16	14	0.14	1973, 1980
	Fraser to Hope	"	5	32	32	0.053	1973, 1980
Great blue heron	Fraser estuary	egg/wet	6	62	62	10.6	1977-78

Notes:
* excludes data from an immediate spill area (up to 75 000 μg/g)
** excludes one value of 620 μg/g
*** Brooksbank, 1983

values in each of these data sets. This underscores the need to establish sampling programs which are statistically based (as well as other requirements) in order to be able to answer the questions of geographic and temporal trends.

The balance of the data cited in the Garrett (1983) report is related to spill incidents and discharges, and to monitoring the immediate sites and neighbouring environments for short periods, usually not more than two years. Collectively, they give a picture of considerable amounts of PCBs in the system although the PCB levels in species common to other parts of Canada (i.e., sucker and rainbow trout) appear to be somewhat lower, certainly than those found in the Great Lakes Region.

A number of "incidents" have occurred in British Columbia, as they have elsewhere. During the period 1979-82, 68 such incidents were reported (Garrett, 1983), most of them involving small amounts of PCBs, albeit of potential significance locally. Several larger spills have occurred. A spill of approximately 1200 kg occurred at Trail in 1976 with potential for pollution of the Columbia River (Garrett, 1983). While some of the contaminated soil (with up to 4% PCB) was removed, the remainder was left since it was felt that leaching/surface runoff would not be a problem in this area of "low" rainfall. It is hoped that monitoring of appropriate samples (not water) from the nearby (700 metres) Columbia River has been undertaken to verify this.

At McLeese Lake, transformers from a hydro sub-station were buried for approximately four years (1972-76) and then removed (Garrett, 1983). This resulted in groundwater contamination (up to 0.201 μg/L in local wells) as well as levels in surface water (up to 0.422 μg/L) and fish (up to 0.595 μg/g muscle). Sampling continued for two years after the incident (four years for soil) by which time the levels had dropped to those typical of other "normal" areas.

Another major spill (approximately 1200 kg) occurred at a pulp mill at Port Edward near Prince Rupert in 1977 (Garrett, 1983). Six months after the spill to the harbour, levels of PCBs in invertebrates were still high but these fell off over the following two years. Concentrations in biota in Chatham Sound, to which the harbour gives access appeared to be largely unaffected.

A number of other discharges were reported in the Garrett (1983) report. Of 25 industrial sites, all except those from a paperboard/packaging plant in the Burnaby area deposited negligible loadings to their receiving waters. Losses from identified effluents at this location showed a significant decrease over the 1979 to 81 year period -- from a daily loading of roughly 100 g to a few grams. A number of sewage treatment plants were also investigated. Only those at Annacis and Iona Islands (roughly 40 g/d and 100 g/d, respectively) were found to discharge significant amounts.

Apparently, no data exist for PCBs in environmental samples from the Yukon part of this Region although some exist for small spill-related situations. Since it has been demonstrated that PCBs are contaminating nearby areas of the Northwest Territories, it is reasonable to assume that they are also present in the Yukon. Recorded storage and use of PCBs in the area amounts to 2019 L (estimated at 3 tonnes).

3.7 National Environmental Trends

Several references are found in the literature concerning Canadian national trends for PCB residues. Vermeer and Peakall (1977), in the earliest of these, found that PCB residues were highest in cormorant, gull and tern eggs from the Great Lakes. The ratios for British Columbia: Prairies: Great Lakes: Gulf of St. Lawrence were 1:1.3:5.5:4.2 for Double-crested Cormorants, 1:13:48:7 for Herring Gulls, and -:2:44:1 for Common Terns.

In human adipose tissue, a qualitatively similar pattern to that of the birds appeared although the extremes were not as great (Task Force, 1976; see Table 1). Adipose tissue collected in a 1976 survey (Mes et al., 1982) subsequent to that reported by the Task Force illustrated the same national pattern although there was a 20 to 30% increase in concentrations for both men (to 1.21 µg/g) and women (to 1.30 µg/g). An investigation of the same type of sample material in 1979-81 in Ottawa and Kingston, Ontario indicated a further 100% increase to an average level of 2.5 µg/g (Williams et al., 1984) although the sample base was not strictly comparable.

In a survey of human milk from across Canada in 1975, a slightly different distribution was observed (Mes and Davies, 1979). In this case, as with adipose tissue, Ontario was highest followed by Alberta and British Columbia; Quebec levels were slightly lower, and Manitoba, Saskatchewan and the Maritimes were the least, each having approximately the same levels. The mean national breast milk concentration at that time was 0.012 µg/L. In a more recent survey (1981 samples, Mes personal communication), Quebec had the highest levels in whole breast milk; other regions were comparable except Manitoba and Saskatchewan which were lower than other areas. The national average was 0.022 µg/L, an 80% increase from 1975.

It is apparent that contamination is greatest in the Great Lakes followed by the Quebec and Atlantic Regions and then British Columbia and the Prairies; the rank of the territories is unknown. What should be apparent is that PCBs are ubiquitous throughout Canada and that all organisms, including man, have been exposed to them. This latter fact suggests a common mechanism for such exposure -- atmospheric transport and deposition is one such possibility.

4 HUMAN EXPOSURE TO PCBS IN CANADA

There are potentially three major routes by which humans are environmentally exposed to PCBs -- through the air they breathe, the water they drink, and the food they eat. Absorption through the skin is an additional possibility but this would be predominantly an occupational exposure. It is possible that skin contact may occur in recreational (such as swimming) or spill situations but since there is inadequate information on these possibilities, they are not considered in this report which deals only with environmental exposure.

4.1 Air

In the preceeding sections, it has been shown that atmospheric contamination by PCBs is widespread. PCBs are found in all parts of the country as indicated by the rain data. In 1984, the rain from such widely separated places as New Brunswick, northern Saskatchewan and the middle of Lake Superior consistently showed levels ranging from 0.001 to 0.003 µg/L. These concentrations indirectly reflect the levels of PCBs in air and it is therefore surmised that vapour concentrations are also similar throughout the same areas.

PCB levels of 2 to 6 ng/m^3 found in air in a variety of Ontario urban areas may be slightly high for use as background levels for non-urban areas (NAS, 1979). An estimate of 1 to 2 ng/m^3 has recently been made for ambient air in the Great Lakes (Eisenreich and Johnson, 1983) and 1 ng/m^3 by Doskey and Andren (1981). If the lowest of these levels is taken as a conservative level for Canada, the exposure of an individual via the atmosphere can be estimated as 0.021 µg/day per adult or 7.6 µg/year (tidal volume* of 500 mL and 29 breaths per minute). Using a higher urban level would raise the estimated level of exposure but the importance of this exposure route relative to food would not be altered.

4.2 Water

An even lower estimate for human exposure is obtained when drinking water is considered. PCBs are seldom found dissolved in water at detectable levels. When they have been reported, it is generally for raw water supplies or surface waters and it is uncertain whether filtration has taken place or whether it has been effective at removing the fine particulate to which the PCBs are often attached. Recent data for open water have been obtained using very large (200-L) centrifuged samples comparable to filtration through a 0.2 µm sieve (McCrea et al., 1985). These data seem to indicate that 0.0002 to 0.0008 µg/L is a realistic level for dissolved PCBs to be found both in the relatively unpolluted waters of the Hudson Bay lowlands (McCrea et al., 1984) as well as the more polluted waters of Lake Ontario (Stevens et al., 1985). Swain (1983) has also estimated similar values for Lake Huron, and Murphy (1984) for Lake Superior. Assuming a "worst case" scenario where none of the dissolved PCBs are removed during the water treatment process plus a daily consumption of two litres, the human exposure from this source would be estimated to be 0.0016 µg/day or an annual intake of 0.58 µg.

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* Tidal volume - volume of air drawn into the lungs during a normal breath.

The assumption of lower water concentrations is in contrast to higher levels that have often been reported for the Great Lakes area, particularly for the Niagara River. Some of these high levels have been determined using methods which should result in estimates of "dissolved" PCBs. PCBs adsorbed to particulate matter should be removed during properly operated water treatment processing; therefore, the drinking water should not be a significant human PCB vector. If the older, higher levels are true measures of the dissolved portion of PCBs, and if, in a "worst case" scenario, they are not removed in the treatment, then communities using such raw water supplies might be exposed to roughly 20 times the preceeding estimate (or 11.6 µg/yr). The higher raw water concentrations were the consequence of failing to remove the fine particulate matter in analyzing the samples much of which would be removed during normal water treatment.

In a study on three wells from the Ottawa region, LeBel and Williams (1980) found a PCB concentration of 0.00013 µg/L (although this was in an area of suspected contamination). This suggests that groundwater contamination in the area is not extensive, even when PCB migration to groundwater is a possibility.

4.3 Food

In contrast to the estimates for air and water, human exposures via food are much higher. Fish from the Great Lakes contain mean whole body concentrations of 0.1 to 8 µg/g and the fillet portions for the larger fish have been estimated to be approximately 50% of that (M. Whittle, Fisheries and Oceans Canada, personal communication). Using a mean value of 2 µg/g for whole fish, the edible portion would contain 1 µg/g (lower levels apply to smelt, but a higher proportion of the fish is consumed). This level certainly represents the "worst case" scenario for freshwater fish in Canada as a whole, since, as indicated by the data in Tables 2, 5 and 9 to 12, levels elsewhere are much lower. Fish from the St. Lawrence River upstream of Montreal, where levels were comparable to those in the Great Lakes, pose roughly the same exposure. A single fish meal of 120 grams at this 1 µg/g level would therefore result in an exposure of 120 µg, a figure far exceeding even the yearly intake by both water and air. Estimates of annual freshwater fish consumption (0.21 kg/yr; Statistics Canada, 1985) would suggest an average exposure of 210 µg/yr or 0.58 µg/d. Some of the total population, particularly the native peoples, have a higher than normal consumption. Reliable data for current levels of PCBs in marine fish are not available. The average national consumption of such saltwater fish is 5.9 kg/yr and a mean level of 0.1 µg/g might be considered to be representative of PCB concentrations in edible portions --based on older data (see Table 1) and similar considerations to those used in the freshwater case. This would result in an estimate of 1.6 µg/person (adult) per day or 590 µg/yr.

The estimation of exposure of the general population through other foods is more difficult, largely because of the limited data on PCB levels in foods. While there are some data for milk, poultry, and meat (Frank et al., 1975, 1979, 1983, 1985a and 1985b) there are little or none for plant products of any sort. There is a small amount of information in Garrett (1983) which deals, in part, with food in the vicinity of McLeese Lake, B.C. where buried transformers lost PCBs to a tributary. Available data on PCB concentrations in different types of foods are found in Table 13.

From the data in Table 13, it appears that the exposure of Canadians has improved substantially for some food types since 1970. This conclusion is based, not only on the few data points presented here, but upon a number of similar surveys spanning the 1970's and into the early 1980's included in the references of Frank et al. There is little evidence to support the contention that root crops take up PCBs from contaminated soils in a manner that leads to human exposure. Carrots have been reported to do so (Iwata and Gunther, 1976) as have beets and turnips (Sawhney and Hankin, 1984) but it appears that most of this is lost when the vegetables are peeled. Leaf and fruit crops have also been reported to take up PCBs (Suzuki et al., 1977); however, Fries and Marrow (1981) and Weber and Mrozek, Jr. (1979) have shown that most of this comes from volatilization from the soil and that the PCBs re-volatilize from the leaves and fruit quickly. Lawrence and Tosine (1977) have reported concentrations of 0.05 to 0.08 µg/g in corn and 0.07 to 0.13 µg/g in grass grown on sludge-treated soil. Buckley (1982) has reported that there was little translocation to foliage of PCBs from "natural" soils near PCB-contaminated sites in the United States and that the PCBs found there were the consequence of atmospheric deposition. Information is not available, however, to indicate general levels

TABLE 13 PCB CONCENTRATIONS IN VARIOUS FOODS

Sample Type	No. of Samples	PCB Concentration (µg/g fat)	% Fat in Food*	Estimated Food Concentration** (µg/g)	Collection Year(s)	Reference
Milk fat	?	0.08	?	0.0030	1970	Frank et al., 1975
" "	?	0.11	?	0.0042	1973	Frank et al., 1975
" "	308	0.035	3.8	0.0013	1977	Frank et al., 1979
" "	359	0.024	3.8	0.00091	1983	Frank et al., 1985a
Beef	150 (av.)	-	-	0.0050	1974	Saschenbrecker, 1985
	150 (av.)	-	-	0.0027	1985	Saschenbrecker, 1985
Beef fat	137	0.25	(20)	0.05	1969-70	Frank et al., 1983
" "	197	0.0095	(20)	0.0019	1981	Frank et al., 1983
Pork	150 (av.)	-	-	0.0096	1974	Saschenbrecker, 1985
	150 (av.)	-	-	0.0037	1985	Saschenbrecker, 1985
Pork fat	35	0.33	(30)	0.099	1969-70	Frank et al., 1983
" "	38	0.003	(30)	0.0009	1981	Frank et al., 1983
Poultry	150 (av.)	-	-	0.0061	1974	Saschenbrecker, 1985
	150 (av.)	-	-	0.0029	1985	Saschenbrecker, 1985
Chicken fat	19	0.95	(17)	0.16	1969-70	Frank et al., 1985b
" "	50	0.0048	(17)	0.00082	1981-82	Frank et al., 1985b
Egg fat	30	0.41	(17)	0.070	1969-70	Frank et al., 1985b
" "	31	<0.01	(17)	<0.002	1981-82	Frank et al., 1985b
McLeese Lake						
Whole milk	2			0.002	1976	Garrett, 1983
Whole egg composites				0.29/0.040	Nov./Dec. 1976	Garrett, 1983
Chicken muscle	1			0.13	1976	Garrett, 1983

Note: * values in brackets are general fat percentages taken from STATSCAN (1985) or Corneliussen (1972)

** presumes all PCBs are in the fat

in Canadian crops grown on "un-amended" soils. It is unfortunate that more attention has not been paid to this (levels of PCBs in plant foods) important topic.

Statistics Canada (1985) lists the daily consumption of a variety of foods in the diet of the average Canadian and the fat content of these foods where appropriate. The average consumption rates for foods which may contain PCBs are: beef (105 g/d), pork (76 g/d), poultry (63 g/d), eggs (34 g/d) and milk (341 mL/d). Using the most recent concentrations shown in Table 13, an estimate of total exposure from these sources is 1.1 µg/d or 400 µg/yr for the "average Canadian". A further 0.6 µg/d or 210 µg/yr is added through the consumption of freshwater fish and possibly 1.6 µg/d or 590 µg/yr for saltwater fish. The totals from all indicated sources might therefore be 3.3 µg/d or 1.2 mg/yr. These estimates must be viewed with caution, however, since some of the contributing values have been determined from fat samples rather than edible tissue and the conversions may not be entirely appropriate. Even where edible tissues were analyzed, the PCBs may not reach the consumer due to loss during preparation. Another caveat, already mentioned, is that the estimate does not include any contribution from fruit and vegetables. This is of particular concern because of the relatively high volume of such material consumed by the average Canadian (233 kg/yr). Even a low concentration can make an appreciable input to the human exposure. In the U.S., surveys which included analyses of plant materials concluded that the intake per person over the 1976 to 1979 year period varied from trace to 0.027 µg/d (Gartrell et al., 1985). As in Canada, this was derived mainly from dairy products, fish, meat and poultry.

Nursing infants, one group particularly at risk, have been studied by Mes and Davies (1979). They reported a mean, Canadian whole human milk concentration of 0.012 µg/g in 1975 and a similar national distribution as was found in human adipose tissue. In a 1981 survey, Mes (personal communication) observed a national average of 0.022 µg/g with the province of Quebec exceeding this by 36% and Manitoba and Saskatchewan being approximately the same percentage below the average. It appears that for the few OECD countries where the data base warrants comparison (Jensen, 1983), Canadian levels are below the average. Generally, levels in these countries seem to have fallen by approximately 50% since the mid-1970's although this trend is not certain. In Sweden, where extensive studies of this sort have been made, levels of PCBs in Stockholm mothers' milk actually rose from 0.014 to 0.030 µg/g (Westoo and Noren, 1978) over the 1967 to 1977 year period and decreased to roughly 0.025 µg/g by 1980 (Jensen, 1984). If the 1981 Canadian level is presumed to have been maintained, an average exposure estimate for this group would be 13.2 µg/d or 4.8 mg/yr, based on a consumption of

600 mL/d. Since this intake is for an infant (estimated 5 kg), its significance on a whole body weight basis (i.e., dose rate) is certainly greater than the same intake for an adult. It is also an exposure at a time in the life cycle which may be particularly susceptible to PCB toxicity.

It appears that the major exposure of adult humans is via the consumption of fish, milk, meats and poultry. Nursing infants are indirectly exposed to the same sources although their intake is more certain. The average daily known adult intake of 3.3 μg, when transformed to a body weight basis for an average 70 kg adult, is 0.047 $\mu g/kg{\cdot}d^{-1}$; for the nursing infants, the estimated intake of 13.2 μg/d would result in a dose of 2.6 $\mu g/kg{\cdot}d^{-1}$ for a weight of 5 kg. For persons such as native peoples who may include a much higher proportion of freshwater fish in their diet, the corresponding exposure for an adult could be higher than 120 μg/d or a dose level of 1.7 $\mu g/kg{\cdot}d^{-1}$.

5 TOXIC EFFECTS OF PCBS

The effects of PCBs on various organisms likely to be affected in Canada was not dealt with extensively in the Task Force (1976) report. This, as much as anything, was the consequence of the terms of reference for the Task Force. This review presents information gathered from several recent reviews (Peakall, 1985; HWC, 1985; D'Itri and Kamrin, 1983; Barros et al., 1984; EPA, 1980; Roberts et al., 1978; IARC 1978). While studies cited in this report are largely more recent ones, it should be remembered that unlike "exposure", the "effects" of PCBs are not a function of when they were determined (other than an improved ability to detect them) and are as valid in 1985 as when many of them were determined in the early 1970's. A caveat to this is the concern that some of the early symptoms described may have been caused by polychlorinated dibenzodioxin and furans known to be present as contaminants in early PCB formulations. This is probably the case for some of the Yusho* symptoms and may have been in other studies pre-dating the mid-1970's.

5.1 Mutagenicity/Carcinogenicity/Teratogenicity

PCBs have been investigated for mutagenic activity using several strains of *Salmonella* sp. bacteria, none of which gave any signs of mutagenicity with or without activation (Schoeny et al., 1979; Shakim et al., 1979). Mutagenicity from PCBs has also been examined using human lymphocyte cells and Aroclor 1254 (Hoopingarner et al., 1972); the fruit fly *Drosophila melanogaster* using PCB mixtures of 30% and 50% chlorine (Nilsson and Ramel, 1974), and _in vivo_ with rats using Aroclor 1254 and 1242 (Green et al., 1975); none of these systems showed signs of chromosomal abnormalities or other evidence of mutagenic activity by PCBs. It is generally agreed that these compounds are not mutagenic although a lower chlorinated mixture (Aroclor 1221) and 4-chlorobiphenyl have been shown to be mildly so (Wyndham et al., 1976). The Wyndham study, however, has not been replicated in other laboratories.

There is little evidence to show that PCBs are carcinogenic, although the U.S. Environmental Protection Agency considers them to be potential carcinogens (EPA, 1980). An epidemiology study by Brown and Jones (1981) using nearly 2600 workers from the electrical industry indicated a slight increase in death from rectal and liver cancers but the statistical significance of attributing this to their occupational exposure has been questioned (Gaffey, 1983). Bertazzi et al. (1981), cited in Silbergeld (1984), reported an increase in the rates of digestive track cancers among Italian workers.

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* Yusho - Japanese word for oil; term given to an incident in Japan when several persons consumed cooking oil contaminated with PCBs.

In animal studies, Ito et al. (1974) observed liver nodules and hepatomas in rats fed PCBs over a 32-week period. Conflicting with this, no increase in tumours were found in a subsequent study with Aroclor 1254 (NCI, 1978) although the nodules were observed. Kimbrough and Linder (1974), also using Aroclor 1254, observed results similar to those of Ito et al. with both rats and mice. In yet another, long-term exposure study with rats exposed to a hexachlorobiphenyl (approximately 60% chlorine), increased liver lesions including nodules and carcinomas were observed (Weltman and Norback, 1979). PCBs are generally considered not to be initiators of tumour growth although there is evidence that Aroclor 1254 can act as a promoter (Preston et al., 1981).

PCBs are reported to have only low teratogenic potential for causing gross deviations in morphology (Welsch, 1983) and reports of such changes are few. There is, however, some evidence which suggests that PCBs are teratogenic in this and other ways. In birds, Carlson and Duby (1973) and Tumasonis et al. (1973) found visible malformations with the chicken using Aroclor 1242 and 1254. Studies with mice and a specific hexachlorobiphenyl have shown malformation of fetuses (NHW, 1985) and cleft palates have also been reported (Watanabe and Sugahara, 1981; Marks et al., 1981) under similar exposure conditions. In humans, small birth size, lowered I.Q. and impaired motor development have been observed for Yusho victims (Harada, 1976). Jacobson et al. (1983) state that PCBs cause all types of teratogenic effects in humans except effects on morphology.

5.2 Birds

Most of the data on birds were formulated from the effects of concentrations of PCBs in their feed over prolonged periods of time. Peakall (1985) has recently reviewed this topic and information from this and other sources appears in Table 14 (emphasis has been placed upon the more sensitive responses).

Most of the data in Table 14 are for the chicken because of its accessibility as a test organism and its economic importance. Fortunately for the purpose of environmental protection, similar responses are observed in other species and, generally, the chicken is among the more responsive of the birds. The reproductive cycle is the most commonly reported life stage which is impaired by PCBs and birds appear to have a dietary threshold somewhere below 5 to 10 µg/g; egg production and hatchability are the

TABLE 14 PCB EFFECTS ON BIRDS

Dose Level (μg/g)	Exposure Period	Effect	Reference
Chicken			
0.1 (diet)	?	reduction in enzymes,	Srebocan et al., 1977
2. "	?	chick growth reduced,	Lillie et al., 1974
2. "	14 w*	reduced egg production,	Platonow and Reinhart, 1973
5. "	?	fertility malformations,	Carlson and Duby, 1973
10. "	?	sexual behavioural effects,	Ahmed et al., 1978
10. "	7 d**	hepatic enzyme increase,	Turner and Green, 1976
10. "	?	reduced egg production,	Scott, 1977
10-15 "	?	hatchability deformities	Tumasonis et al., 1973
Quail			
0.1 (diet)	?	enzyme changes, porphyria	Vos et al., 1971
5. "	12 w	enzyme increase	Dieter, 1974
American kestrel			
0.5 (diet)	14 d	increased metabolic rate	Bird et al., 1983
Pheasant			
12.5 μg/g(bw)/d	17 w	reduced egg production	Dahlgren and Linder, 1971
Ring dove			
10. (diet)	12 w	inattentiveness to eggs	Peakall and Peakall, 1973
10. "	?	embryotoxic, damaged chromosomes	Peakall, 1972 Peakall et al., 1972

* week
** day

effects most often reported. Eggshell thinning leading to poor reproductive success, seems to be caused by factors other than PCBs. This has been established for the mallard (Risebrough and Anderson, 1975), the ring dove (Peakall, 1971) and the chicken (Lillie et al., 1974).

While PCBs do not appear to be mutagenic, they are apparently teratogenic to a number of bird species. Some of the relevant studies cited in Peakall (1985) involved direct injection into the egg and, while this is difficult to relate to natural exposures from the parent, it serves to indicate that PCBs are potent teratogens to chickens (Carlson and Duby, 1973; Tumasonis et al., 1973; Cecil et al., 1974) and ring doves (Peakall et al., 1972).

In addition to reproductive effects in birds, a number of reports refer to the PCB alteration of enzyme levels, usually those of the liver. The lowest of such levels reported for chicken and quail, occurred at 0.1 μg/g in the feed (Srebocan et al., 1977). Another study on the American kestrel noted an increase in the metabolic rate with a concentration of 0.5 μg/g PCBs in the diet (Bird et al., 1983). These effects are probably reversible and are not necessarily adverse in themselves to the well-being of the individual birds. They are, however, symptomatic of the bird under stress and consequently, PCB dosages causing stress should be considered adverse.

Results from laboratory testing for PCB effects alone may not be entirely appropriate in the field where a multitude of chemicals act upon the bird -- either additively, synergistically, or antagonistically. It is virtually impossible to resolve this dilemma. There are effects observed in the field which are attributed to PCBs mainly because these compounds are among the more prominent contaminants found in carcasses of birds or their eggs. Mineau et al. (1984) discuss several of these effects in relation to herring gull monitoring in the Great Lakes. Such effects are similar to those presented in Table 14, namely reduced reproduction due to smaller egg clutches and poor hatching and survival of chicks. From an environmental protection viewpoint, it should be assumed that these effects are due to PCBs. This should in no way preclude assuming that the effects are due to some other compound; it is merely the equivalent of the "worst case" scenarios from exposure.

5.3 Plants

There are relatively few reports on the effects of PCBs on plants themselves. This probably arises from their apparent insensitivity to most organochlorine substances. An illustrative study was reported by Strek et al. (1981) who found no significant growth effects on harvested corn or sorghum grown in soil with a PCB-concentration up to 1000 μg/g, and the growth of soybean and beet was inhibited only at 1000 μg/g PCBs in the soil. They also found that there was little translocation of PCBs from a point of application on a leaf into the rest of the plant although loss of the radio-label suggested volatilization (Weber and Mrozek, Jr., 1979). Fescue also showed little ability to take up PCBs.

There are a few studies, however, which give some cause for re-evaluating the premise that plants are insensitive to PCBs, at least in aquatic systems. In fresh water from Saginaw Bay, Lake Huron, McNaught et al. (1984) demonstrated a 0.005 µg/L threshold for the inhibition of photosynthesis by a hexachlorobiphenyl using a natural phytoplankton community. In another study, Powers et al. (1982) observed that 50 µg/L added as adsorbed PCBs (Aroclor 1254) on clay sediments was available to phytoplankton and caused reduced photosynthesis to a natural estuarine community. If these authors' sorption constant of approximately 10 000 is used, an exposure concentration of 0.005 µg/L can be estimated. Conner (1981) has found that this response (photosynthesis) is the most sensitive plant response to PCBs.

Other investigations in marine and estuarine ecosystems have noted effects at much higher levels of PCBs. O'Connors et al. (1978) found that as little as 1 µg/L PCB would suppress photosynthesis, cell biomass and chlorophyll-a in a natural phytoplankton community and also affected diatoms; there was also mortality to a copepod at this level. Biggs et al. (1978) observed that 5 µg PCB/L inhibited algal growth and additionally, affected long-term species composition. Harding, Jr. (1976) found effects at 10 µg/L with marine phytoplanton as did Kleppel and McLaughlin (1980).

5.4 Mammals

A large number of reports exist on the effects of PCBs on mammalian species. As noted previously in the section on birds, only the more sensitive responses are noted in this report and representatives of these for mammals appear in Table 15. Most of this material was obtained from D'Itri and Kamrin (1983); Barros et al. (1984); and NHW (1985).

PCBs have been tested with primates - notably monkeys. These animals show many of the symptoms that have been observed in humans following accidental exposures either in "Yusho-type" incidents or in occupational situations. It is difficult to generate any statistical significance and confidence that is usually obtained with other species because relatively small numbers of these animals are used in any particular experiment. Acknowledging this, it is noted that with exposures as short as four weeks, serious effects including mortality are found with diet levels of two tetrachlorobiphenyl isomers (approximately 49% chlorine) as low as 0.3 µg/g (McNulty et al., 1980). The correspondence of this to a dose level is uncertain but an estimate of 8 to 12 kg as a weight and 180 to 200 g/d as a food intake (Villeneuve, pers. com.) results in an effect level at approximately 0.006 µg/g body weight per day.

TABLE 15 TOXIC EFFECTS IN MAMMALS DUE TO SUBLETHAL EXPOSURES TO PCBs

Exposure Level*		Exposure Duration (weeks)	Effects	Reference
Diet ($\mu g/g$)	Dose ($\mu g/g{\cdot}d^{-1}$)			
Monkey				
0.3	(0.006)	4 to 24	chloracne, weight loss, death, thymus involution, hyperplasia of gastric mucosa	McNulty et al., 1980
2.5	(0.05)	72	lactation transfer, reduced birth size, post-exposure effects	Allen et al., 1980
2.5	(0.05)	24	reduced birth size, acne, abortion	Barsotti et al., 1976
2.5	(0.05)	24	chloracne, lowered gamma-globulin	Thomas and Hindsill, 1978
5.	(0.1)	72	non-viable births	Allen et al., 1980
(5 to 20)	0.1 to 0.4	136 to 152	immunosupression, stillbirth, impaired development	Truelove et al., 1982
(100)	2.	10 to 17	MFO induction	Iverson et al., 1982
(200)	4.	1 menstrual cycle	decreased estrogen, anovulatory cycle	Mueller et al., 1978
Mouse				
0.16	(0.03)	2.5	placental and lactation transfer	Masuda et al., 1979
0.3	(0.05)	4	enlarged liver	Biocca et al., 1981
(5.9)	1.	1.5 during pregnancy	liver discolouration, kidney nodules	Marks et al., 1981
(11.8)	2.	1.5	malformed fetus, cleft palate	Marks et al., 1981
	0.025 mg/d	9 to 10	lengthened oestrus cycle, decreased implantations	Orberg and Kihlstrom, 1973
	0.5 mg/animal	6/7	reduced implantation, MFO induction	Orberg, 1978
Rat				
5.	(0.3)	4	reduced thyroid development, litter size	Collins and Capen, 1980
5.	(0.3)	5	increase liver lipid, reduced thyroid development	Kaza et al., 1978
(5.)	0.3	4	immunosupression, reduced spleen and thyroid development	Van Velsen et al., 1984
(17.)	1.	1/7	increased weanling growth, poor reproductive development	Gellert, 1978
Mink				
0.7	(0.07)	3	reduced reproduction, kit deaths	Hornshaw et al., 1983
3.6	(0.36)	15	100% mortality	Platonow and Karstad, 1973
5.	(0.5)	35	deaths, complete reproductive failure	Bleavins et al., 1980

* - values in parentheses are conversions of diet levels to doses at the rate of 10 μg/g (diet) = 0.6 μg/g (bw)/d for the rat; 10 μg/g (diet) = 1.7 μg/g (bw)/day for the mouse; 10 μg/g (diet) = 1 μg/g (bw)/d for mink; and 10 μg/g (diet) = 0.2 μg/g (bw)/day for the monkey

Mice are also affected by PCBs but are influenced at somewhat higher levels than monkeys. While not an effect as such, transfer of PCBs to fetuses and nursing mice occurs at low levels (0.16 µg/d (diet), equivalent to approximately 0.03 µg/g (body weight per day) of exposure in the diet (Masuda et al., 1979) and liver enlargement has been observed at dietary concentrations only slightly above this (Biocca et al., 1981). Similar effects, although at higher levels, occur with rats as well.

Mink is another mammalian species that is sensitive to PCBs. Exposure of these animals to Great Lakes fish, a normal food source, and to contaminated meat indicated that they are extremely affected by the toxicity of PCBs. Dietary concentrations as low as 0.6 µg/g resulted in reduced reproduction and some deaths among kits (Hornshaw et al., 1983; Ringer, 1983). For an assumed body weight of 1.5 kg and a daily food intake of 150 g, this exposure indicates dose effects at 0.07 µg/g (body weight per day).

Most of the other studies available in the literature report effects at much higher levels of exposure than those presented here. They do show, however, that PCBs affect mammals particularly their reproduction and early development.

5.5 Fish and Aquatic Organisms

The toxicological effects of PCBs on aquatic organisms has been as well studied as the effects on terrestrial species. Unlike their air-breathing counterparts, however, exposure data for aquatic organisms are mainly in the form of PCB concentrations in water rather than actual doses received by the species under study. This manner of reporting fosters the presumption that effects on such creatures are derived from direct exposure or uptake from water; food is seldom considered in laboratory studies with fish and lower trophic level organisms. Also, since laboratory systems seldom represent the true complexity of the real environment, these two factors cause difficulty in extrapolating laboratory data to the field.

Much of the information available on the effects of PCBs is derived from studies of acute exposures to these compounds (i.e., exposures over relatively short periods or even a single dose). These effects occur at levels greatly above those which are actually found in the environment and they often involve death as the observed end-point. The focus of this review is on effects arising from long-term (chronic) exposures to sublethal levels of PCBs. An extensive compilation of all such PCB effects on aquatic organisms is found in the U.S. Environmental Protection Agency's "Ambient Water Quality Criteria" (EPA, 1980). Additional review material can be found in a paper by

Laake (1984) given at the PCB seminar held in den Haage, Netherlands in September, 1983. Aquatic organisms also have different sensitivities to temperature which can be a factor in the succession in algal communities (Fisher and Wurster, 1973) and in fish behaviour (Peterson, 1973). Some of the more sensitive responses to PCB exposures in water are presented in Table 16.

While some of the effects in Table 16 may be reversible, others are not. Even reversible effects, however, can cause loss of succession in a community and as such, they are adverse effects. There are very few studies using natural communities other than those using the lower trophic levels. Investigations using marine (or brackish water) plankton systems indicate that responses of such communities are as sensitive as most of the organisms listed in Table 16. A reasonable conclusion would be that a chemical stress level (lethal or otherwise) to a laboratory organism would also have an adverse effect on the organism in its natural environment at comparable concentration levels.

An examination of the nature of the effects in Table 16, together with others from uncited literature, indicates many of the same types of biological responses occur in both the aquatic and terrestrial worlds. Thyroid effects have been reported for the rainbow trout (Leatherland and Sonstegard, 1979), results observed with other species include behavioural (avoidance) effects (Hansen et al., 1974); impaired structural development (bone) (Mauck et al., 1978); liver changes (Hacking et al., 1977; Freeman et al., 1982); testicular abnormalities (Sangalang et al., 1981); and enzyme induction (Willis et al., 1980; Addison et al., 1982; Melancon and Lech, 1983). These are important in the survival of the terrestrial species; there is no reason to believe they are less important for their aquatic counterparts.

A point of some concern is the consideration of the medium as the only exposure route for aquatic organisms. Bioaccumulation of PCBs by aquatic species is a well-documented phenomenon; it is the sum of the bioconcentration (the accumulation of body residues directly from the water medium) and biomagnification (accumulation through the food chain). The report by EPA (1980) lists a wide range of bioaccumulation factors (ratio of the concentration in whole body to the concentration in water) most of which were derived under circumstances which would measure mainly bioconcentration. For freshwater organisms, these bioconcentration factors range from 2 700 to 274 000 with those for fish occupying the upper end of the range (saltwater species had similar values). These factors, when applied to an estimate of 0.001 μg/L for the Great Lakes and elsewhere in Canada, would result (for an average factor of 100 000) in whole body residues of 0.1 μg/g. That much higher levels are observed for some predator species

TABLE 16 EFFECTS OF PCBS ON FISH AND OTHER AQUATIC ORGANISMS

Species	Exposure		Observed Effect	Reference
	Concentration ($\mu g/L$)	Duration*		
Freshwater				
Lake trout	0.05	50 d	- increased mortality rate	Seelye and Mac, 1981
Mosquitofish	0.1	1.5 h	- avoidance	Hansen et al., 1974
Fathead minnow	0.4	240 d	- larval death	Defoe et al., 1978
Fathead minnow	0.9	4 m	- ATPase inhibition	Cutkomp et al., 1972
Brook trout	0.43	128 d	- backbone development	Mauck et al, 1978
Largemouth bass	1.5	8 d	- "early life stage" LC_{50}	Birge et al., 1979
Rainbow trout	2.	8 d	- "early life stage" LC_{50}	Birge et al., 1979
Alga (*S. quadr.*)	0.1	24 h	- reduced carbon fixation	Luard, 1973
Midge (*T. diss.*)	0.65/0.45	3 w	- larvae/pupae LC_{50}	Nebeker and Puglisi, 1974
Daphnia magna	0.48	2 w	- reproductive impairment	Nebeker and Puglisi, 1974
Mosquito larvae	1.5	?	- inability to metamorphose	Sanders and Chandler, 1972
Saltwater				
Sheepshead minnow	0.14	28 d	- reduced reproduction	Hansen et al., 1973
Sheepshead minnow	0.93	21 d	- embryo/fry LC_{50}	Schimmel et al., 1974
Diatoms (*R. setig.*, *T. pseud.*)	0.1	8 d	- inhibit growth	Fisher and Wurster, 1973
Phytoplankton (*T. pseud.*;*I. galbana*)	1.	?	- reduce cell division	Harding and Phillips, 1978
Protozoan	1.	4 d	- reduce growth and population density	Cooley et al., 1972
Phytoplankton community	1.	4 d	- reduced biomass and mean size	O'Connors et al., 1978
Zooplankton community	1.	4 m	- composition and diversity	Hansen, 1974
Oyster	1.	4 d	- reduce rate of shell growth	Duke et al., 1970
Pink shrimp	1.	15 d	LC_{50}	Nimmo and Bahner, 1976

* d - day; h - hour; m - month; w - week.

would seem to indicate an additional source of PCBs; their food would be a logical suspect. High residue levels could also have arisen from higher dissolved PCB levels in the water but this is not believed to be the case (see Chapter 4).

Great Lakes whole body residues in fishes are dangerously high when compared to dietary effect levels which have only been determined for a few freshwater fish. Rainbow trout fed at 0.39 μg/g (diet) for 30 days were found to suffer extensive mortality and deformation of fry (Hogan and Brauhn, 1975); coho salmon had stimulated thyroid activity at 0.42 μg/g for 72 hours (Mayer et al., 1977); and, bluegills showed inhibition of ATPase when fed at 0.6 μg/g (Desaiah et al., 1972). These effects, even the biochemical ones, will undoubtably affect the organism's ability to maintain itself in its community.

6 HUMAN HEALTH EFFECTS

There are a number of reports that deal directly with the effects of PCBs on humans. A chapter by Gaffey (1983) and the review by NHW (1985) outline the epidemiological studies that have been undertaken. The most famous occurrence was a contamination of cooking oil by Kanechlor 400 (approximately 48% chlorine) in southern Japan in 1968 which caused what is known as "Yusho" or oil disease. A variety of symptoms were observed - chloracne, pigmentation changes and gastrointestinal upset were among the immediate effects (Kuratsune et al., 1972); death was also reported. Some of these effects persisted for as long as six years after the ingestion of the contaminated cooking oil with the age group from 13 to 29 years being particularly affected. There were longer lasting effects as well. School children exposed were observed to have retarded growth (Yoshimura and Ikeda, 1978) and there were adverse effects on fat metabolism (Higuchi, 1976). Blood enzyme levels were changed, the immune system suffered dysfunction as did the lungs, liver and ovaries of affected persons (Kuratsune and Shapiro, 1984). Most of these latter effects, however, can not be statistically established as due to the PCBs; chlorinated dibenzodioxins and furans were also present as impurities and are believed to be responsible for some of the observed effects (Kashimoto et al., 1981).

A similar incident occurred in Taiwan in 1979. Concentrations of impurities in the contaminated oil there were lower (at least an order of magnitude lower) and mean PCB blood levels of 49 µg/L were found nine months to one year after the original exposure (Chen et al., 1980). Similar acne and pigmentation symptoms were observed as were found in Japan (Wong et al., 1982); evidence of porphyria was also observed (Chang et al., 1980).

One set of recent studies that requires comment because of its relevance to environmental exposure is the investigation of the effect of Lake Michigan fish in the diet of pregnant and nursing mothers. Jacobson et al. (1984) and Fein et al. (1984) have noted various behavioural and developmental defects in newborn children born to Michigan mothers who were consumers of fish contaminated with several substances, prominent among which were PCBs. Freshwater fish consumption by these mothers (at 6.7 kg/year), was considerably higher than the value computed for Canadian annual consumption (0.21 kg/yr - Statistics Canada, 1985) and the degree of contamination is undoubtably greater for the Michigan fish. The observation of these symptoms does not directly implicate these compounds but it does provide presumptive evidence. These PCB effects are being investigated further.

A variety of symptoms associated with PCB exposure have been identified from a number of studies of occupational situations. Silbergeld (1984) listed cancer, neurotoxicity and reproductive toxicity in addition to those effects already mentioned. In a study of 2567 workers at a PCB transformer plant in the United States (Brown and Jones, 1981), only rectal and liver cancer appeared to be slightly increased, the significance of which has been questioned (Gaffey, 1983). A variety of neurological symptoms were reported by Fischbein et al. (1979) in an examination of capacitor plant workers. Human sperm density was adversely affected by PCB and other contaminants with the strongest correlations occurring with levels of higher chlorinated PCB congeners (Dougherty et al., 1980). These few reports do not exhaust the literature on effects and reference is made to the review papers already mentioned. A number of reports qualitatively indicative of the effects found with human PCB exposures are given in Table 17, along with mean reported blood levels and type of PCBs, where available. A summary report of occupational exposure is provided by Kuratsune and Shapiro (1984).

TABLE 17 EFFECTS ATTRIBUTED TO OCCUPATIONAL EXPOSURE OF HUMANS TO PCBS

Observed Effect	PCB Type (Aroclors)	Blood Levels	Reference
Chloracne, liver disorder	N/A*	N/A	Meigs et al., 1954
Dermatitis, eye irritation	1242	58 to 282 (μg/L)	Ouw et al., 1976
Liver metabolic changes	N/A	N/A	Alvares et al. 1977
Chloracne, altered liver and blood biochemistry	N/A	33 (μg/L) (5.6 μg/g fat)	Chase et al., 1982
Dermatitis, neurological symptoms, liver enzyme induction	1254, 1242 (mainly)	33 to 227 (μg/L)	Fischbein et al., 1979
Chloracne, altered blood biochemistry	N/A	N/A	Hara et al., 1975
Pigmentation, liver enzyme induction, cholesterol	N/A	370 (μg/L)	Hasegawa et al., 1972
Chloracne	N/A	130 to 520 (μg/L)	Inoue et al., 1979
Dermatitis	N/A	820 (μg/g)	Kitamura et al., 1973
Dermatitis, liver enzyme induction	1242, 1254	342 (μg/g)	Maroni et al., 1981
Dermatitis, liver and blood biochemistry, neurological symptoms	1242, 1016	89 to 502 (μg/L)	Smith et al., 1982
Rectal and liver cancer	1254, 1242, 1016	N/A	Brown and Jones, 1981
Mortality	1254	N/A	Bertazi et al., 1981

* N/A = Not Available

7 ASSESSMENT

Hazard evaluation of a chemical in the environment is done to determine the danger that the chemical presents to both man and other organisms. This danger is determined by two aspects of the chemical --its exposure to the organism (including man), and the effects that it has upon the organism. Exposure, in turn, is determined by the properties of the chemical; the characteristics of the system in which the exposure is occurring; the duration of the exposure; and the numbers of the particular organism being exposed. In a strict sense, it is also a function of the internal physiology of the exposed organism. Effects are, of course, dependent on the organism considered. For environmental purposes, attention is usually focussed upon the more sensitive organisms in the system being evaluated, and usually upon the most sensitive life stage of these organisms.

Each assessment feature, exposure, and effects, can be determined in the field or predicted with the help of laboratory data. Since cost and concern over deliberately polluting the environment are important factors, laboratory data are frequently employed unless a spill or other accidental exposure and effect information becomes fortuitously available. The use of laboratory data has some risk associated with it since the environment, with all its complexity, may increase or attenuate the effect of a chemical on any one of its component parts. It should be obvious that actual environmental concentrations and effects are preferred when they are available. It must also be accepted that assessments must frequently be made and regulations developed without such 'definitive' evidence.

The hazards presented by PCBs in the environment are described in this report. For human health purposes, material was gathered from the Bureau of Chemical Hazards, Health and Welfare Canada. There is no attempt to define what an acceptable level of risk for PCBs should be since this depends upon a value judgement and there is presently no acceptable methodology for assigning value to aesthetic aspects of the environment. Rather, a comparison is made of the exposure concentrations which have or are occurring in Canada with the effect levels which have been determined (or predicted) for relevant Canadian organisms residing in the compartments of the ecosystem for which exposure estimates are available. The process is called hazard assessment.

7.1 Aquatic Organisms

It is the contention of this report that PCBs are transported extensively via the atmosphere and that the load there is probably a dynamic one turning over frequently.

This being the case, it is possible that most of the country receives comparable surface loadings of these compounds, altered by local inputs but at comparable magnitudes. In the Great Lakes, with their large volumes, areas, and retention times, the material is retained in the system while still remaining available for accumulation by resident biota - either directly from the water or indirectly via the food web. It is this accumulation in particular which has given rise to the environmental concern there.

Levels of PCBs in fish of the Great Lakes ecosystem have fallen somewhat from those reported in the Task Force Report. Dissolved water values of less than 0.001 μg/L for Lake Ontario (considered to be the most polluted in this system) may be an artifact of new procedures employed in analysing large samples and any presumption of a decreasing trend in water should therefore be further investigated. The upper limit for levels of dissolved PCBs in the lower Great Lakes is estimated to be approximately 0.001 μg/L and concentrations in the St. Lawrence should be about the same. Limits in other parts of Canada could be one or more orders of magnitude lower based on levels in biota other than humans. However, because there has been little direct determination of water levels in other locations and because human adipose tissue concentrations do not differ as extensively as do residues in other organisms, it is recommended that the 0.001 μg/L level be considered as a "worst case scenario" for water in Canada. Table 16 indicates no toxic effects to aquatic organisms that come closer than a factor of fifty times this level, although the effect observed at this level was death. More information on sublethal effects are required for sensitive species (such as lake trout). The information on human exposure presented in Chapter 4 indicates that there is little concern for humans from such levels.

There are, however, concerns about the indirect accumulation of PCBs. In the Great Lakes, as elsewhere, there are residues in the tissues of aquatic organisms which serve as food for other organisms. The levels of PCBs in the recipient organisms accumulate, possibly to toxic levels but concentrations vary greatly. A minimum estimate of whole body residues for fish and fish-eating birds would be at least 0.1 μg/g based on the biota listed in Table 6. This is coincidentally close to that which would be estimated using the above water level and a bioaccumulation factor of 100 000. The ratio of the lowest dietary effect level for aquatic organisms (rainbow trout) to this tissue level is only four times and the effects observed include mortality. This "safety margin" is small and should be cause for considerable concern. Many fish species in this system have whole body concentrations well above this level. Since many of the effects are subtle and difficult to distinguish from those caused by other stresses, it can only be assumed that effects are being felt by predator species at almost every trophic level. In their 1974 report, the Water Quality Objectives Subcommittee (WQOS, 1975) opted to provide a second line of defence against PCBs by defining a maximum fish tissue level intended to protect consumers of Great Lakes fish. While it has not reduced the exposures to anywhere near the level, it has at least served to identify the problem.

7.2 Terrestrial Organisms

Unlike the aquatic environment, there are three major routes that terrestrial organisms can come in contact with PCBs -- in the air they breath, the food they eat and the water they drink. This, at least, is the case for the mammalian and avian species. For humans, dermal contact may be an occupational concern but this is not considered here.

Effects on soil insects are not well known. An evaluation of the loadings of PCBs that are likely to be deposited from rain (assuming a Canadian-wide mean of 0.004 μg/L in the rain and an annual precipitation of 75 cm) and allowing a factor of three times for vapour phase transfer, leads to the conclusion that some 10 $\mu g/m^2 \cdot yr^{-1}$ could be deposited. This amount would only increment the concentration of the top 1 cm by 0.0002 μg/g annually and deeper layers correspondingly less. Direct exposure of most terrestrial organisms to such levels is probably not a concern although it may be in areas utilizing sewage sludge. Although worms live in and eat a medium which receives atmospheric depositions, there is inadequate information on the effects of PCBs in these organisms.

The vital volumes of air for terrestrial organisms other than humans are not well documented. On the assumption this is dependent on body weight (perhaps more true for mammals than for birds), it is unlikely that breathing is a major exposure route for terrestrial organisms. As estimated for man from data in Chapter 4, this mechanism appears to represent only about 2% of the total exposure estimated from fish and other food.

It appears that food is the major cause for the various tissue residues that have been found in the Canadian environment. For both the avian and non-human mammalian species, there are virtually no data on PCB levels in their food sources. An exception to this would be commercial mink and the herring gull both of which subsist substantially, at least in the Great Lakes area, on fish. Tables 14 and 15 indicate that the lowest effect level, albeit probably reversible enzymatic ones, occurs in the range of

0.1 to 0.5 µg PCB/g (diet) for birds and 0.2 to 0.3 µg/g for mammals. Such effects may influence the ability of the organism to cope with its environment and should therefore be considered detrimental. Fish in the Great Lakes area and other areas such as the coastal region of British Columbia, the St. Lawrence River region of Quebec, and possibly marine mammals in parts of the Arctic region contain PCBs at levels above these effect levels and regular consumers of them must be considered to be at some risk from PCBs.

Plant grazing animals may also be at risk. While the amount of PCBs deposited by precipitation is small on an areal basis, it may well form a high concentration on the surface of leaves and grasses. It undoubtably volatalizes from such surfaces but is also re-deposited. The steady-state levels on such material are unknown but it is suspected to be significant since grazing animals do accumulate PCBs in their tissues and milk at levels which are probably responsible for the human adipose tissue levels of the Canadian population. The shortage of data on residues in wildlife and birds is a major difficulty in making conclusions concerning the hazard of PCBs to terrestrial organisms.

It is not possible to estimate the precise PCB-intake for humans because various food types are consumed and residue levels in some food types are not known. The daily intake of 3.3 µg/g for adults estimated in Chapter 4 can be expressed as a dose level (for a 70 kg adult) of 0.047 µg/kg body weight per day which, compared with the lowest reported dietary effect level for monkeys (the most PCB-sensitive mammalian species) of 6 µg/kg body weight per day, represents a safety factor of approximately one hundred and thirty times. There is some margin of safety, therefore, for the average Canadian but it should be borne in mind that this level of protection is based on effect levels. The "no effect" level and that margin of safety may be somewhat less. Effect levels are, of course, dependent on intake from fish which may be much higher among native peoples -- possibly enough to reach levels at which effects have been observed in experimental animals. There is also no account taken of possible intake from fruit, vegetables, and cereals.

In the case of nursing infants, the calculated exposure is 2.6 µg/kg body weight per day. Although this exceeds the Canadian tolerable daily intake of 1 µg/kg body weight per day, this latter value was based on calculations for a lifetime exposure at this level. Exposed infants would normally only encounter these exposure levels for three months on average and seldom beyond one year. The significance of the exposure is therefore uncertain.

8 CONTROL OF PCBs IN CANADA

8.1 Inventory

An inventory of PCBs in Canada has recently been published by Environment Canada (Environment Canada, 1985). This document indicated that a total of about 24 300 tonnes or about 16.2 x 10^6 litres of PCBs have been accounted for across Canada. Tables 18 and 19 summarize this inventory information by industry sector and application. Total imports of PCBs into Canada up to 1977, when their importation was greatly restricted, were estimated to be 4 X 10^4 tonnes (Environment Canada, 1985); no record exists of PCBs being manufactured in Canada. To date, therefore, approximately 40% of all PCBs brought into Canada can no longer be accounted for. This is about 1.6 X 10^4 tonnes (10.5 x 10^6 litres) of PCBs which may already be dispersed in the environment.

TABLE 18 PCB INVENTORY BY APPLICATION FOR CANADA (Environment Canada, 1985)

Application	Quantity (tonnes)												
	B.C.*	Alta.	Sask.	Man.	NWT	Ont.	Que.	N.B.	N.S.	PEI	Nfld.	Total	% Total (Canada)
Electrical													
Capacitor	378.3	94.9	32.8	320.6	14.1	649.4	1188.5	69.1	31.0	0.3	14.6	2793.6	11.5
Transformer	1065.4	512.2	563.3	130.9	84.4	7964.4	3549.6	22.9	337.6	13.8	308.3	14 852.8	61.1
Electromagnet	18.5					25.0	11.7	0.7				55.9	0.2
Other					2.5	41.3	31.9		3.7			79.4	0.3
Mechanical													
Hydraulic	9.8					7.7						17.5	0.1
Heat Transfer						0.1						0.1	0.0004
Vacuum Eqpt.						1.0						1.0	0.004
Storage for Disposal**	224.3	5011.7	21.5	17.3	5.8	880.6	233.3	21.3	49.8	0.8	43.6	6510.0	26.8
TOTAL	1696.3	5618.8	617.6	468.8	106.8	9569.5	5015.0	414.0	422.1	14.9	366.5	24 310.3	
% Total (Canada)	7.0	23.1	2.5	1.9	0.4	39.4	20.6	1.7	1.7	0.1	1.5		

* Includes Yukon Territory

** Includes on-site and commercial storage

Note: totals differ slightly from Table 19 due to number rounding.

TABLE 19 PCB INVENTORY BY INDUSTRY SECTOR FOR CANADA (Environment Canada, 1985)

Industry Sector	Quantity (tonnes)												
	B.C.[a]	Alta.	Sask.	Man.	NWT	Ont.	Que.	N.B.	N.S.	PEI	Nfld	Atlantic[c]	Total	% Total (Canada)
Cement	50.5	35.0	7.6	20.9		130.0	35.4	40.3	1.2		0.7		321.6	1.3
Chemical Manufacturing	14.4	74.2	2.3	0.1		164.4	83.2					23.6	362.2	1.5
Electrical	3.3	0.1	4.4	5.4		265.3	40.7	0.1	0.1				319.4	1.3
Food/Beverage	28.1	19.4	20.1	5.2		228.5	77.6	13.7	3.6	4.7	0.8		401.7	1.7
Forestry/Pulp/Paper	737.5	79.2	0.6	47.6		704.5	667.6	210.2	229.2		79.1		2755.5	11.3
Government Institutions	37.2	47.2	17.5	22.0	88.1	537.7	509.0	8.7	52.6	2.6	21.4		1344.0	5.5
Hospital/School	16.0	112.5	10.7	5.9		643.7	185.0	2.7	23.2	4.2	26.6		1030.5	4.2
Mining/Smelting	247.2	5.0	335.9	27.7	16.1	775.9	459.0	6.0	10.8		72.6		1956.2	8.0
Petroleum	4.6	81.8	25.0	2.6	1.3	49.7	24.1		5.8			7.1	202.0	0.8
Refining Metal	12.5	2.3				313.9	87.1						415.8	1.7
Rubber		12.0				74.3	46.9						133.2	0.5
Steel/Iron	5.5	34.8	10.3	6.1		1891.6	259.9	3.4	48.9		4.7		2265.2	9.3
Textile			0.1			90.0	107.2		6.8				204.1	0.8
Utility	482.5	108.5	179.5	298.9	1.3	2212.8	1392.2					278.6	4954.3	20.4
Other[d]	57.0	6.9	3.8	26.3		1217.2	869.6	7.2	15.0	1.4			2204.4	9.1
Commercial Storage		5000.0[b]				269.5	170.5						5440.0	22.4
TOTAL	1696.3	5618.9	617.8	468.7	106.8	9569.0	5015.0	414.0[e]	422.1[e]	14.9[e]	366.6[e]		24 310.1	
% Total (Canada)	7.0	23.1	2.5	1.9	0.4	39.4	20.6	1.8	1.7	0.1	1.4			

Notes:

a includes Yukon Territory

b this storage figure includes casings and waste material such as cleanup and dirt; of this total, approximately 1160 tonnes has been identified as being present in drums; it is unknown as to the quantities of PCBs in the remaining total

c this column has been created to list quantities of PCBs in the Atlantic provinces where there are a minimum number of companies in an individual province, in order to maintain confidentiality

d where only one company has been inventoried for the Atlantic provinces in an individual sector, quantities have been added to the "other" category

e actual totals are given for each province, even though data by industrial sector are included in the "Atlantic" column

Most of the PCBs presently accounted for in Canada are either in storage or in use in electrical equipment (see Tables 18 and 19). These potential sources of release are the deserved focus of most of the environmental concern; however, there are other possible sources of PCBs such as those from industrial processes involving:

- dyes/pigments made with halogenated solvents;
- chlorosilane derivatives;
- organophosphorus trichloride derivatives;
- alkyl chlorophosphine derivatives;
- chlorinated naphthalene derivatives;
- chlorinated aryl derivatives;
- chlorobutane derivatives;
- chloroxylene derivatives;
- biphenyl derivatives; and
- chlorinated phosphate esters. (V. Buxton, pers. comm.)

The significance of these sources is unknown and the inadvertent production or loss of such PCBs are not controllable under the Environmental Contaminants Act, nor have they been reviewed in the context of the new Canadian Environmental Protection Act. There may be other products, in addition to those listed, which have an uncontrollable PCB content.

8.2 Releases to the Environment

8.2.1 Landfills. Deposit of PCBs and PCB-containing materials/products/equipment into landfills was the primary method of disposal for these chemicals in North America until the mid 1970's (U.S. NRC, 1972, 1979). Although no specific information or estimates are available for Canada, it is estimated that from 45% to 50% of North American production of PCBs has been placed in landfills and dumps (Task Force, 1976). Similarly, it was estimated that 51% of the 1970 U.S. production of PCBs was disposed of in landfills and dumps (Nisbet and Sarofim, 1972).

The practice of occasionally setting fire to the contents of landfills to reduce the volume of the solid waste (paper, etc.) had unknown consequences on PCBs contained in those landfills. Some destruction of PCBs, production of polychlorinated dibenzofurans and polychlorinated dibenzodioxins, and probably volatilization of PCBs may have occurred but there is no information available to determine the extent of this. It is also possible that leachate from such sites entered the groundwater and subsequently the surface waters. Again, insufficient information exists to further define the problem.

8.2.2 Industrial Effluents. Release of PCBs in the effluents of manufacturing industries which used PCBs was once an important mode of entry for these chemicals into the aquatic environment (U.S. NRC, 1979). However, since PCBs are no longer being manufactured, and new equipment can no longer contain PCBs, this problem has probably been significantly reduced (Webber et al., 1983). At present, the major source of industrial effluent contamination is from leaks and spills from older electrical, hydraulic or other PCB-containing equipment still in use or in storage awaiting disposal (Environment Canada, 1981). In all cases, this equipment will eventually be replaced with PCB-free equipment and this source of contamination should be eliminated. Under present regulations, however, replacement will not take place until PCB-containing equipment is gradually phased out, and this type of equipment has a long lifetime.

Two manufacturing processes which have been of concern in terms of PCBs in effluents are the recycling of waste oil and waste paper (Environment Canada, 1981). Waste oil is sometimes contaminated with PCBs when used hydraulic, electrical or other fluids containing PCBs are inadvertently or otherwise combined with oil destined for recycling. The processing of this contaminated waste oil results in the addition of PCBs to the effluents; use of this oil on roads is also a source of PCB release to the environment. Chlorobiphenyl Regulations Nos. 2 and 3 (1985) may do much to alleviate these problems.

PCBs have also been used in the manufacture of printing inks and carbonless copy paper (Nisbet and Sarofim, 1972). When waste paper is processed for recyling, the de-inking and pulping processes result in the release of some of the PCBs to effluent streams (Garrett, 1983). PCBs released in this manner, however, are primarily associated with suspended solids in the effluents and improved effluent treatment to reduce suspended solids has resulted in a significant reduction in the amount of PCBs released to the environment (Garrett, 1983). The PCBs, however, are still incorporated in the sludge and may be dispersed to the environment as such. As the PCB content in recycled paper is reduced with increasing time, this problem should eventually be resolved.

8.2.3 Municipal Effluents. A 1973 survey of Ontario municipal wastewaters and treatment facilities indicated that municipal effluents are also a source of PCBs found in the aquatic environment (Shannon et al., 1973). Treatment processes removed an average 56% of the PCBs contained in the influents. Influent PCB loadings were higher for industrialized areas indicating that industrial effluents to municipal sewer systems are probably a significant contributor of PCBs to municipal wastewaters.

In British Columbia, a study of sewage treatment plants (Garrett, 1983) found that PCBs were discharged from sewage plants there. At ten plants, an average 53% of input concentrations were removed during a 1976 survey, although higher mean final concentrations were observed in an independent survey in 1978. At concentrations of <1 µg/L in the effluent, the levels from the latter survey are comparable to those of the secondary plants examined in the Ontario study (Shannon et al., 1973). As in the Ontario study, the sludge was the main repository of the PCBs.

8.2.4 Industrial and Municipal Wastewater Treatment Sludge. A further problem exists with PCBs in municipal and industrial wastewater treatment sludges. Disposal practices for sludges include incineration, dumping at sea, and disposal on land (landfills and spreading on agricultural land, sludge farms, etc.) (Webber et al., 1983). PCB concentrations in sludges from Ontario municipalities were measured as high as 76.6 ppm (dry weight) (Shannon et al., 1973); those from B.C. had a maximum of 2.5 ppm (Garrett, 1983). Using this material for landfilling or application to agricultural land can result in the contamination of soil and groundwater. Disposal at sea can result in marine water and sediment contamination; and potentially affect marine life and the quality of fish or shellfish consumed. Incineration, if inefficient, can result in the contamination of the atmosphere with volatilized PCBs and conversion products of polychlorinated dibenzofurans and dioxins.

Tests of soil where sludges were applied to land demonstrated a build-up of PCBs in soils, with higher PCB levels generally being associated with greater sludge applications (Webber et al., 1983). PCBs have been detected in the scrubber water of a sludge incineration unit (Shannon et al., 1973) indicating that PCBs were volatilizing from the incinerated sludge. Past landfilling of PCBs, in general, has resulted in extensive groundwater contamination as seepage and leaching from the landfill site occur (Pupp, 1985).

8.2.5 Industrial Leaks and Spills

A program to monitor major leaks and spills of hazardous wastes and materials was established by Environment Canada in 1972. Leaks or spills of PCBs, often occur during the transport of obsolete electrical or other equipment containing PCBs, and to date, about 340 spills and leaks of PCBs have been recorded. Even assuming no cleanup, these have amounted to the loss of approximately 89 tonnes of PCBs which represents less than 1% of the total amount of PCBs estimated to have been dissipated in the Canadian environment. While leaks and spills are relatively minor sources of PCB release, they are of great significance in the vicinity where they occur and are often the manner in which the public's attention is focussed on these compounds. They may also be a more significant fraction of the current losses. The high profile of spills has resulted because these incidents often occur in and around densely inhabited areas where they result in relatively high levels of contamination in a small area with potential for human exposure.

Efforts to reduce the occurrence of spills during transport of PCB-containing equipment have recently been put into effect through regulations under the federal Transport of Dangerous Goods Act (TDGA) and corresponding provincial legislation. Other measures to reduce loss of PCBs by leakage and spills during storage of these chemicals have also been implemented.

8.3 Destruction Technology

Environment Canada is investigating the feasibility of implementing an accelerated phase-out of PCB-containing equipment (Dillon, 1984). An accelerated phase-out was recommended by the PCB Board of Review in its report of March, 1980 in which the Board recognized that natural attrition did not promise a sufficiently early resolution to the environmental PCB problem (Board of Review, 1980). Central to the phase-out strategy, however, is the availability of suitable PCB destruction facilities. Such destruction facilities do not presently exist in Canada, and other countries possessing them are not accepting imported PCBs for destruction at their facilities.

A variety of destruction technologies are available which could provide safe and efficient destruction of PCBs. Some of these technologies are reviewed in two 1983 reports (Environment Canada, 1983; Ackerman et al., 1983), while more recent developments and advances in these technologies are being investigated by federal and provincial agencies. These technologies include rotary kiln and multiple hearth incineration (which can handle both solids and liquids); chemical plasma processes; molten salt processes; fluidized bed processes; and plasma arc processes; most of which are restricted to liquids. There are also pyrolysis systems which may have both the throughput capacity as well as the necessary destruction efficiency. Thermal conditions of 1200°C with two to five second contact time and suitable mixture with air or oxygen are required for efficient incineration.

8.4 Federal Regulatory Controls of PCBs

Introduction of a regulation to limit the use of PCBs was the first step toward controlling PCBs in Canada. In 1978, Chlorobiphenyl Regulation No. 1 was promulgated under the Environmental Contaminants Act. This regulation prohibited PCBs from any "open" uses but continued to permit existing "closed" (largely electrical) applications which were in use in Canada before March 1, 1977. The regulation also allowed the continued use of PCBs in the manufacture of electrical capacitors and transformers.

Amendments to Chlorobiphenyl Regulation No. 1 were put into effect in 1980. As a result of these amendments, existing PCB-containing electrical equipment could continue to be used, but no new PCB-containing equipment could be manufactured (including electrical equipment), and no new PCBs could be used to refill or otherwise service existing equipment. As of 1980, therefore, no new PCBs could be put into service in Canada. Also included in these amendments were restrictions prohibiting the use of PCB-containing equipment in the handling of produce, food or animal feed, or anything intended to be added to food or animal feed. These latter restrictions were intended to protect human health which could be endangered if PCBs became incorporated into the food supply through leaks and spills during food processing.

Two additional Chlorobiphenyl Regulations came into effect in August, 1985. Chlorobiphenyl Regulation No. 2 (Product) stipulates that any product offered for sale in Canada (new or used) cannot be contaminated with or contain PCBs at a concentration greater than 50 ppm by weight. Exceptions to this are for equipment whose purpose is the destruction of PCBs, or for storage awaiting destruction, or when the equipment is an integral part of a building, plant or structure. This regulation is intended to speed up the phase-out of PCB-containing equipment in accord with the Board of Review's recommendation.

Chlorobiphenyl Regulation No. 3 (Release) effective August 1, 1985 was designed to reduce the deliberate release of PCBs to the environment. This regulation sets two restrictions on the release of material containing PCBs: 1) 50 ppm is the maximum concentration of PCBs by weight that can be released to the environment by any commercial, manufacturing or processing activity except for use as dust suppressants on unpaved roads where the limit is 5 ppm; and 2) 1 gram per day is the total allowable release in the course of operating, servicing, maintaining, decommissioning, transporting or storing of PCB-containing equipment.

"Regulations Respecting the Handling, Offering for Transport, and Transportation of Dangerous Goods" have been promulgated under the Transportation of Dangerous Goods Act and took effect on July 1, 1985. These regulations cover all hazardous goods and wastes, including PCBs. The requirements for transportation of PCBs include: PCBs or equipment containing PCBs must be enclosed in a rigid leak-proof container during transport; or large articles (transformers for example) must be drained of PCBs if they cannot be properly enclosed. All containers or equipment must be secured to the transport vehicle during transport, a direction which applies to equipment containing PCBs at a concentration greater than 50 ppm by weight. The regulations also specify procedures and requirements for the proper marking, identifying, classifying, packaging, handling, and inspecting during transport. These make the documentation (manifesting) of the movement of PCBs during interprovincial or international transport mandatory and prescribe specific responses in the case of accidents or spills of hazardous waste during transport. It is hoped that this latter requirement will improve the efficiency and effectiveness of spill response and cleanup activities. Training programs for the transportation industry and for inspectors are underway. It is yet to be seen if these more stringent requirements for transport of PCBs will have any impact on the numbers and volumes of PCB spills across Canada.

Other regulations, guidelines and action levels that exist for PCBs both in Canada and elsewhere in the world may be compared in Table 20.

TABLE 20 RECOMMENDATIONS AND GUIDELINES FOR THE LIMITATION OF PCB EXPOSURE

	Region	Concentration	
Air			
Ambient	Ontario	35 ng/m^3 (1-yr av.) 150 ng/m^3 (24-h av.) 450 ng/m^3 (0.5-h av.)	
	New York State	1.0 µg/m^3 (short-term max.)	
Occupational	Canada	42%Cl	1 mg/m^3 (over 8 h); 2 mg/m^3 (over 15 min.)
		54%Cl	0.5 mg/m^3 (over 8 h); 1 mg/m^3 (over 15 min.)
	British Columbia	42%Cl	1 mg/m^3 (over 8 h); 2 mg/m^3 (over 15 min.)
		54%Cl	0.5 mg/m^3 (over 8 h); 1 mg/m^3 (over 15 min.)
	Alberta	42%Cl	1 mg/m^3 (over 8 h); 2 mg/m^3 (over 15 min.)
		54%Cl	0.5 mg/m^3 (over 8 h); 1 mg/m^3 (over 15 min.)
	Quebec	42%Cl	1 mg/m^3 (over 8 h); 2 mg/m^3 (over 15 min.)
		54%Cl	0.5 mg/m^3 (over 8 h); 1 mg/m^3 (over 15 min.)
	New Brunswick	42%Cl	1 mg/m^3 (over 8 h); 2 mg/m^3 (over 15 min.)
		54%Cl	0.5 mg/m^3 (over 8 h); 1 mg/m^3 (over 15 min.)
	Newfoundland	42%Cl	1 mg/m^3 (over 8 h); 2 mg/m^3 (over 15 min.)
		54%Cl	0.5 mg/m^3 (over 8 h); 1 mg/m^3 (over 15 min.)
	Ontario	0.05 mg/m^3 (indoor-av. for 40-h work wk) 450 ng/m^3 (outdoor-max. half hour av. for destruction facilities)	
	U.S. ACGIH	42%Cl	1 mg/m^3 (over 8 h); 2 mg/m^3 (over 15 min.)
		54%Cl	0.5 mg/m^3 (over 8 h); 1 mg/m^3 (over 15 min.)
	U.S. NIOSH	1 µg/m^3 (avrage over 40-h work wk)	
	West Germany	42%Cl	1 mg/m^3 (8-h av. assumed)
		54%Cl	0.5 mg/m^3 (8-h av. assumed)
	East Germany	1.0 mg/m^3 (8-h av. assumed)	
	Sweden	0.5 mg/m^3 (8-h av. assumed)	
	Australia	1.0 mg/m^3 (8-h av. assumed)	
	Czechoslovakia	42%Cl	1.0 mg/m^3 (8-h av. assumed)
		54%Cl	0.5 mg/m^3 (8-h av. assumed)
	Romania	42%Cl	1.0 mg/m^3 (8-h av. assumed)
		54%Cl	0.5 mg/m^3 (8-h av. assumed)
	Finland	42%Cl	1.0 mg/m^3 (8-h av. assumed)
		54%Cl	0.5 mg/m^3 (8-h av. assumed)
	Japan	1.0 mg/m^3 at 25°C and 1 atm. (8-h av. assumed)	
Water			
Surface	Manitoba	0.002 µg/L	
	Ontario	0.001 µg/L	
	Quebec	0.001 µg/L	
	IJC Great Lakes	0.001 µg/L	
	U.S. EPA	0.00079 µg/L (cancer rate 1:10^5) 0.000079 µg/L (cancer rate 1:10^6) 0.0000079 µg/L (cancer rate 1:10^7)	
	U.S. EPA	0.014 µg/L (freshwater)	
	U.S. EPA	0.03 µg/L (saltwater)	

TABLE 20 RECOMMENDATIONS AND GUIDELINES FOR THE LIMITATION OF PCB EXPOSURE (Cont'd)

	Region	Concentration
Recreational	Quebec	0.1 μg/L
Drinking	Saskatchewan	undetectable (desirable) (3 μg/L (max.)
	Ontario	3 μg/L (max.)
	Quebec	0.1 μg/L (recommended standard)
	Nova Scotia	3 μg/L (max.)
	U.S. EPA	0.125 mg/L (1-day SNARL) 0.0125 mg/L (10-day SNARL)
	U.S. NAS	0.35 mg/L (1-day SNARL) 0.05 mg/L (10-day SNARL)
Sediment		
Dredged spoil maximum for open water dumping	Ontario Great Lakes	0.05 mg/kg
	Quebec	0.1 mg/kg
	Canada	1 mg/kg
	Holland, Norway, Sweden	1 mg/kg
Soil		
Considered contaminated	Quebec	5 ppm
Target level for cleanup	Saskatchewan	5 ppm
Surface Contamination		
Target level for cleanup	Quebec	2.5 mg/cm^2 (Arochlor 1242) 1.25 mg/cm^2 (Arochlor 1254, 1260)
Solid Waste/Sludge		
Maximum for farm application	U.S. FDA	10 mg/kg (ppm-dry wt)
Solid Waste		
Considered hazardous	Canada	>50 ppm
	Quebec decontamination or destruction	>50 ppm or leachate >0.01 mg/L
	sanitary landfilling	≤15 ppm or leachate ≤0.005 mg/L
	secure landfilling	>15 to 50 ppm or leachate 0.005 to 0.01 mg/L
Liquid Waste—Hazardous		
	Canada	50 ppm
	Quebec	0.3 ppm
Liquid Waste—Effluents		
	Canada	50 ppm
	Ontario	5 μg/L (mobile destruction facilities)
Liquid Waste—Road Oil		
	Canada	5 ppm (max.)
	Quebec	0 ppm
Liquid Waste—Storage Loss		
	Canada	1 g/d per piece of equipment

TABLE 20 RECOMMENDATIONS AND GUIDELINES FOR THE LIMITATION OF PCB EXPOSURE (Cont'd)

	Region	Concentration
Food — Tolerance Levels		
-fish	Canada	2 mg/kg (edible portion)
-milk/dairy products	Canada	0.2 mg/kg (fat basis)
-poultry	Canada	0.5 mg/kg (fat basis)
-eggs	Canada	0.1 mg/kg (whole less shell)
-beef	Canada	0.2 mg/kg (fat basis)
-animal feed component	Canada	2.0 mg/kg (fish oil)
-milk/dairy products	U.S. FDA	1.5 mg/kg (fat basis)
-poultry	U.S. FDA	3.0 mg/kg (fat basis)
-red meat	U.S. FDA	3.0 mg/kg
-eggs	U.S. FDA	0.3 mg/kg
-fish/ shellfish	U.S. FDA	2 mg/kg (edible portion)
-infant/ junior foods	U.S. FDA	0.2 mg/kg
-animal feed components	U.S. FDA	2.0 mg/kg
-finished animal feed	U.S. FDA	0.2 mg/kg
-paper food packaging material	U.S. FDA	10 mg/kg
Acceptable Daily Intake		
	Canada	1 µg/kg/day
	U.S. EPA	3.0 µg/kg/day[1] 0.3 µg/kg/day[2]
	U.S. FDA	3.0 µg/kg/day[3] 2.5 µg/kg/day[4] 20 µg/kg/day (50 days)[5] 1.0 µg/kg/day (22 months)[5]

Notes:

1. Based on rat studies with a safety factor of 100;
2. Based on rat studies with a safety factor of 1000;
3. Based on rat studies;
4. Based on dog studies;
5. Based on the Yusho Incident; calculated using an average total exposure level of 2000 mg/adult; since dioxins, furans and quaterphenyls were also present, the estimate of a "no observed effect" level due solely to PCBs is impossible from the Yusho data.

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APPENDIX

REFERENCE SEARCH

APPENDIX: REFERENCE SEARCH

The scientific and other literature was searched during May-June 1985. Data bases accessible through DIALOG and Environment Canada's WATDOC were examined. The search was limited to references from roughly 1975 to the present. The following data bases were searched using the indicated search characteristics. There was some, although not extensive, replication among the different bases; not all references were useful.

Data Base	Search Characteristics	No. of References
CAS-82-85	PCB* or Polychlorinated Biphenyl not (Human or Electric*) and (Measurement* or Fate or Impact* or Effect* or Environment* or Residue* or Behaviour or Behavior) RN = 11096-82-5 or 11097-69-1 or 25323-68-6 or or 25429-29-2 or 26914-33-0	138
CAS-80-81	as for CAS 82-85	82
CAS-77-79	as for CAS 82-85	124
AST	PCB or Polychlorinated Biphenyl* or Tri/Tetra/Pentachlorobiphenyl* or PCB 1254 or PCB 1260	4
CENV	PCB*	463
	PCB* and 1254	60
	PCB* and 1260	9
	PCB* and (Environment* or Effect* or Occurence* or Measur*)	45
	PCB* and (Behaviour* or Fate* or Impact* or Residu*)	68
	Polychlorinated Biphenyl and (Environment* or Effect* or Occurence* or Measur*)	49
	Tetra/Pentachlorobiphenyl	5
DEL	as for AST	37

CAS - Chemical Abstracts
AST - Arctic Science & Technology
CENV - Canadian Environment
DEL - Delft Hydro